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Management of Midwestern Landscapes for the Conservation of Neotropical Migratory Birds

Edited by
Frank R. Thompson, III

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MANAGEMENT OF MIDWESTERN LANDSCAPES FOR THE CONSERVATION OF NEOTROPICAL MIGRATORY BIRDS

Edited by :

Frank R. Thompson, III
North Central Forest Experiment Station
1-26 Agriculture Building
Columbia, MO 65211 USA

Preface:

This proceedings is the result of a symposium held December 5, 1995 in Detroit Michigan. The symposium is part of a series of biennial symposia sponsored by the North Central Section of the Wildlife Society that are held in conjunction with the Midwest Fish and Wildlife Conference.

This symposium was organized in response to interest by members of the North Central Section of the Wildlife Society and land managers in the conservation of neotropical migratory birds and landscape level management. The proceedings is organized as a series of papers beginning with three introductory papers that review threats to Midwestern birds, geographic and ecological distributions of birds in the Midwest, and the importance of metapopulation and source-sink population structure to conservation efforts. The remaining eight papers address the conservation of migrant landbirds in different landscapes and ecosystems in the Midwest.

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Frank R. Thompson, III

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THREATS TO BREEDING NEOTROPICAL MIGRATORY BIRDS IN THE MIDWEST

Scott K. Robinson[1]

ABSTRACT.—Neotropical migrants face threats throughout their complex life cycles. This review focuses on threats to breeding neotropical migratory birds (NTMBs) with emphasis on studies conducted in the Midwest. The two primary threats appear to be habitat fragmentation and habitat loss, which are usually closely associated with each other. Midwestern studies have overwhelmingly shown that many NTMBs of forests and grasslands are area-sensitive and prone to reproductive failure in small, edge-dominated habitat patches. Cowbird parasitism is often identified as the chief cause of reproductive failure, but nest predation may be just as important. Fragmented landscapes appear to serve as population "sinks" for many species that are only being maintained by immigration from "source" populations in large, unfragmented habitat tracts. Direct evidence for source/sink metapopulations, however, is lacking. Habitat loss has been most acute for floodplain forest, grasslands, and savanna habitats, all of which should to be the focus of restoration efforts. Changes in habitat structure and composition resulting from a lack of natural disturbances may threaten some of the Midwest's highest-priority species. Other threats such as global warming, drought, loss of stopover and winter habitat, and decreasing food supply either have not yet been well documented or are largely outside the control of land managers in the Midwest.

An approach that potentially reduces most of the major threats to breeding NTMBs in primarily agricultural landscapes is to conserve and restore large tracts. When combined with effective management (e.g., edge reduction) of networks of smaller tracts, macrosites might help maintain regional populations of NTMBs by providing "source" habitat where reproductive success is high enough to produce a surplus. In primarily (>80 percent cover) forested or grassland landscapes, managers should be most concerned with maintaining a representative mix of habitat types.

INTRODUCTION

Neotropical migratory birds (NTMBs) have become the focus of the largest international conservation effort ever for nongame wildlife that is not yet endangered (Terborgh 1989, Hagan and Johnston 1992, Finch and Stangel 1993, Martin and Finch 1995). NTMBs are an extremely diverse group of more than 250 species that occupy virtually every terrestrial habitat in North America. In some forest communities, NTMBs form up to 90 percent of the breeding bird community (Terborgh 1989). The insects consumed by NTMBs may play a role in maintaining forest health (Marquis and Whelan 1994). Neotropical migrants are being used increasingly as indicators of ecosystem health and examples of the need to manage on a larger spatial scale than previously recognized (Maurer 1993, Villard and Maurer 1996). The "Partners in Flight" program (Finch and Stangel 1993) has become a model of the kinds of interagency cooperation and coordination that will be necessary to address ecosystem-level and global conservation issues.

Conservation of NTMBs, however, is a very complex task because their life cycle makes

[1]*Illinois Natural History Survey, 607 E. Peabody Drive, Champaign, IL 61820.*

them vulnerable to loss of habitat on geographically widely separated breeding, wintering, and migratory stopover habitats (Martin and Finch 1995). Although there is a growing consensus that NTMBs face significant threats throughout the year (reviewed in Askins *et al.* 1990; Finch 1991; Sherry and Holmes 1993, 1995; Robinson 1993), some have argued that threats on the winter grounds are more likely to limit populations than those on the breeding grounds (e.g., Rappole and McDonald 1994, Rappole 1995). The American Midwest, however, has historically provided abundant evidence that loss and fragmentation of breeding habitat poses a severe threat to many species (see table 1). Indeed, many of the studies listed in table 1 formed part of the scientific platform on which the Partners in Flight program was based (Finch 1991). Since the publication of these studies, our knowledge has expanded greatly. A new generation of studies have been completed and we are increasingly able to offer more useful and scientifically rigorous management recommendations (Faaborg *et al.* 1993, 1995; various papers in Martin and Finch 1995 and Maurer and Villard 1996), which is the primary emphasis of most chapters in this book. As will become clear from this paper, however, we still have a great deal to learn about NTMBs and how best to conserve them. Managers must remember that our understanding of basic NTMB population biology is still in its infancy. Any recommendations we make now may have to be modified later as we obtain more data, conduct more experiments, and, above all, have had a chance to monitor impacts of management activities over the long run.

The purpose of this paper is to provide an overview of threats faced by NTMBs with a focus on those most relevant to the Midwest. I have divided these threats into three general categories, each of which has several related sub-issues. The first threat is habitat fragmentation, which I define as the disruption in the continuity of a habitat (Lord and Norton 1990). This definition includes the more extreme case of insularization of habitat patches by agricultural or residential land use and the subtler fragmentation that can occur when human activities such as logging create a mosaic of successional stages within a tract. The second general threat is habitat loss, which usually accompanies habitat fragmentation but is often targeted at specific habitats such as grasslands or floodplain forest. The third general threat is changes in habitat structure and composition, which can result in the loss of species that require particular microhabitats. There are other potential threats such as global climate change (Root and Schneider 1995, Rotenberry *et al.* 1995), but we currently know little about them and I mention them only briefly at the end of the paper. Additional threats posed by tropical deforestation may be very important for some species (e.g., cerulean warbler: Robbins *et al.* 1992; reviewed in Petit *et al.* 1993, Rappole and McDonald 1994, Rappole 1995), but are mostly beyond our ability to affect here in the Midwest (Faaborg *et al.* 1996). Throughout this paper I will cite mostly studies that were conducted in the American Midwest; for more geographically comprehensive reviews, see Askins *et al.* (1990), Askins (1993), Finch (1991), Faaborg *et al.* (1993), Robinson (1993), Robinson and Wilcove (1994), Wilcove and Robinson (1990), and Sherry and Holmes (1993, 1995).

THREATS RELATED TO HABITAT FRAGMENTATION

The Midwest has long provided some of the most robust data demonstrating the severe threat posed by habitat fragmentation (table 1). In this section, I review the major ways in which neotropical migrants are threatened by habitat fragmentation.

Area-sensitivity is the absence of a species from woodlots below a certain size even if suitable habitat is present. Area-sensitivity has been well documented in the Midwest, largely because of the abundance of discrete habitat "islands" in agricultural landscapes (e.g., Bond 1957; Howe 1979, 1984; Ambuel and Temple 1983; Kendeigh 1982; Hayden *et al.* 1985; Howe *et al.* 1985; Blake and Karr 1994, 1987; Herkert 1994a,b). Freemark and Collins (1992) found that area-sensitivity was more pronounced in Illinois than in the more extensive forests of Ontario. Nevertheless, small (10-100 ha) isolated forest tracts in central Illinois often contain many, if not most forest birds as long as there is at least one major ravine system present (Blake and Karr 1984, 1987; Robinson 1988, 1992). Even some forest raptors occupy small tracts in Illinois (Robinson 1991). Area-sensitivity is most pronounced when very small (<10 ha) woodlots are included in the analysis (e.g.,

Table 1.—*Selected Midwestern references relating to NTMBs in chronological order*

Paper (Year)	Summary
Bond (1957)	Area-sensitivity (see also Ambuel and Temple 1983) and habitat requirements of forest birds
Graber and Graber (1963)	Adaptability of many birds to early agricultural landscapes
Faaborg (1980)	Fallacy of managing for local diversity at the expense of regional diversity by promoting habitat heterogeneity (see also Robinson 1988, Reese and Ratti 1988)
Kendeigh (1982)	One of the longest-term censuses ever conducted (see also Brawn and Robinson 1996)
Brittingham and Temple (1983)	Edge effects on cowbird parasitism and link with declining neotropical migrants
Blake and Karr (1984, 1987)	Area-sensitivity in birds of forested islands, see also Hayden *et al.* (1985) and Howe (1984)
Temple and Cary (1988)	Edge effects on parasitism and predation; landscape model
Gibbs and Faaborg (1990)	First paper establishing a link between pairing success and fragmentation
Johnson and Temple (1990)	First paper showing that grassland fragmentation was similar to forest fragmentation in its effects on breeding birds
Robinson (1992)	Extreme effects of fragmentation on nesting success (see also Bollinger and Linder 1994)
Herkert *et al.* (1993)	First management manual for NTMBs of forests and grasslands
Thompson (1993)	Simulated potential impacts from edge effects and habitat composition resulting from timber harvest
Herkert (1994a,b)	Area-sensitivity and habitat requirements of grassland birds
Thompson (1994)	Movement and habitat use patterns of brown-headed cowbirds
Robinson *et al.* (1995a)	Landscape-level scale of fragmentation
Donovan *et al.* (1995a,b)	Modeled source-sink demography in Midwestern populations

Blake and Karr 1987). Area-sensitivity therefore appears to be more extreme in Missouri than it is in Illinois, perhaps a reflection of the lack of alternative larger tracts in Illinois. There are, however, relatively more residents and short-distance migrants in small woodlots (Blake 1983) and the NTMBs in small woodlots are a fairly consistent subset of those present in larger woodlots (Blake 1991). Area-sensitivity is only a threat if birds avoid small woodlots where they could breed successfully. If small woodlots are poor habitat, however, then avoiding small woodlots many be adaptive (see below).

Cowbird parasitism is perhaps the best-documented threat to breeding NTMBs in midwestern woodlots (Brittingham and Temple 1983; Robinson *et al.* 1995a, in press; Thompson *et al.*, in press). For many forest species in the Midwest, levels of parasitism are positively related to levels of forest fragmentation (fig. 1). Cowbird parasitism is related to fragmentation because cowbirds do not feed in most forest or natural grassland habitats (reviewed in Robinson *et al.* 1993, 1995b). Instead, they feed in pastures, row crops, and lawns and commute up to 7 km to areas where they search for nests to parasitize (Thompson 1994; Thompson and Dijak, in press). In Illinois, cowbirds feed in row crops as well as pastures (Thompson and Dijak, in press), which may explain why they are more abundant in some Illinois landscapes than in Missouri landscapes (Thompson 1994; Thompson *et al.*, in press). Landscapes fragmented by agricultural land uses in the Midwest therefore provide ideal conditions for cowbirds, and woodlots in the agricultural Midwest have some of the highest parasitism levels ever documented (Robinson 1992; Robinson *et al.* 1995a,b; Brawn and Robinson 1996; Thompson *et al.*, in press). The ratio of cowbird to host abundance in fixed-radius point counts has proven to be a good predictor of community-wide parasitism levels for forest songbirds in the Midwest (Robinson *et al.*, in press; Thompson *et al.*, in press). Management recommendations to reduce cowbird parasitism have been described in Robinson *et al.* (1993).

Nevertheless, cowbird parasitism is far from a universal problem in the Midwest. The large forest tracts of the Missouri Ozarks and northern Wisconsin have insignificant levels of parasitism (Robinson *et al.* 1995a; Thompson *et al.*, in press; Donovan *et al.* 1995a) and in south-central Indiana, only a few species (e.g., red-eyed vireo and hooded warbler) sustain the kinds of parasitism levels that are likely to create population-level problems for these species (Winslow *et al.*, in press; Thompson *et al.*, in press). Similarly, cowbird parasitism appears not to be much of a problem for grassland birds in Illinois (Kendeigh 1941; Robinson *et al.*, in press), although it may be a more severe problem in Kansas (Elliott 1978) and the northern Great Plains (Koford *et al.*, in press; Davis and Sealy, in press). Edge/second-growth species are also less heavily parasitized than forest species, at least in some areas (Robinson *et al.*, in press). Large-scale reduction of cowbird populations (a difficult task at best: Rothstein *et al.* 1987, Robinson *et al.* 1993) therefore may have little impact on populations of most NTMBs (see also Rothstein and Robinson 1994, Robinson *et al.* 1995b). Local cowbird trapping, however, may be desireable to enhance populations of endangered species (e.g., Kirtland's warbler: reviewed in Robinson *et al.* 1993, 1995b) and to increase productivity in "macrosites" (see below) in landscapes where it is impossible to preserve or restore tracts that are large enough to be free of intense cowbird parasitism (Robinson *et al.* 1995a).

Nest predation also appears to be most severe in fragmented forest landscapes, although the effect is not as pronounced as it is with parasitism (Robinson *et al.* 1995a) (fig. 1). As with parasitism, however, there is much interspecific and inter-habitat variation in levels of nest predation (Robinson *et al.* 1995a). The tremendous diversity of nest predators makes reducing nest predation a complex task (Reitsma *et al.* 1990). Studies of the behavior of key nest predators (e.g., black rat snakes *(Elaphe obsoleta)*: Durner and Gates 1993, Withgott 1995) and of the relative abundances of nest predators in tracts of different sizes will help managers determine how to reduce predation to "acceptable" levels (which have yet to be determined: Martin 1992).

Edge effects.—Studies in the Midwest have provided some of the most widely cited examples of negative effects of edge on nesting success. Levels of nest predation (e.g., Temple and Cary 1988, Johnson and Temple 1990, Burger *et al.* 1994) and cowbird parasitism (Brittingham and Temple 1983, Temple and Cary 1988, Johnson and Temple 1990) have

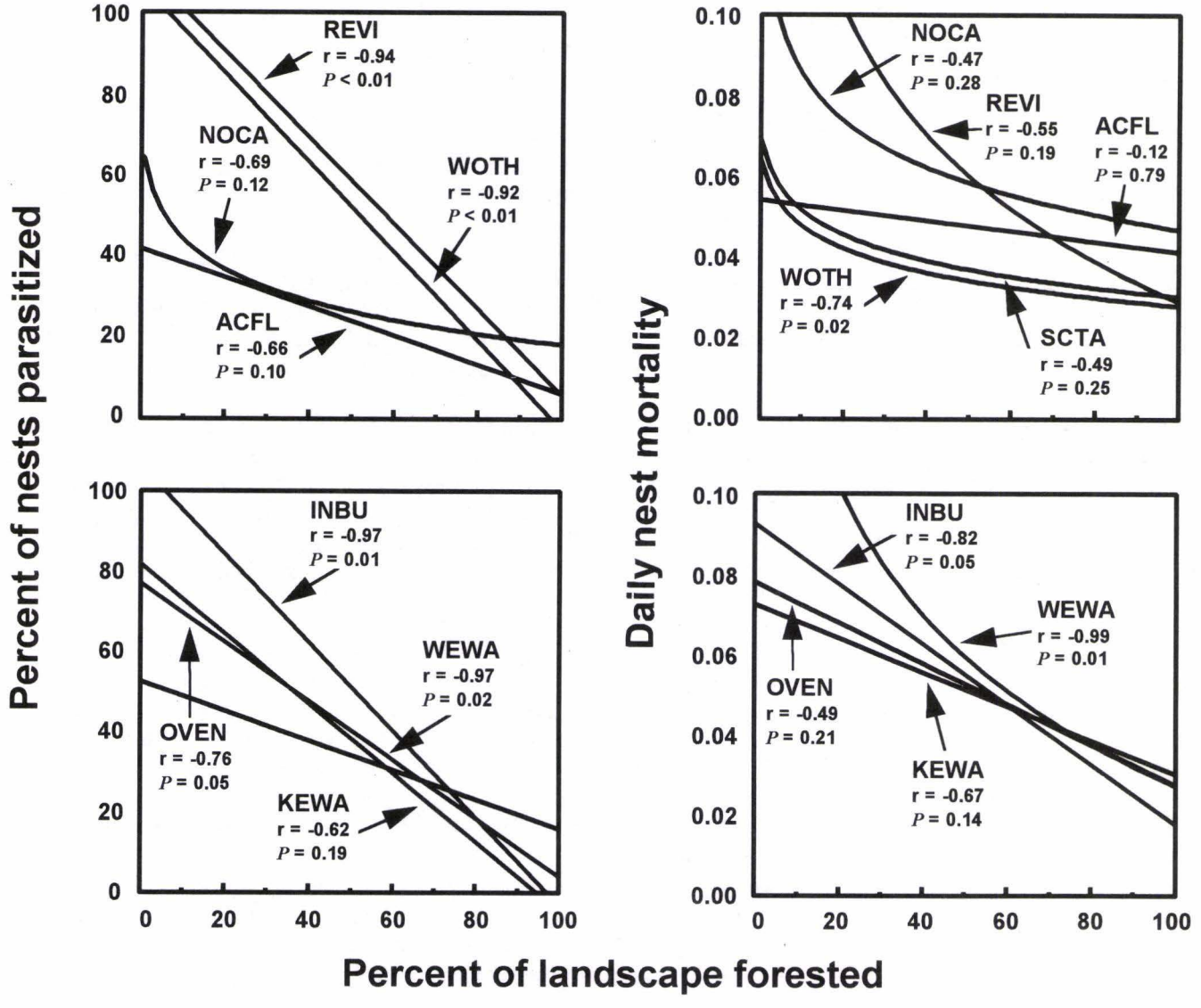

Figure 1.—*Correlations between levels of brood parasitism and nest predation with the percent of forest cover in nine Midwestern landscapes. Abbreviations are: ACFL, Acadian flycatcher; INBU, Indigo bunting; KEWA, Kentucky warbler; NOCA, Northern cardinal; OVEN, ovenbird; REVI, red-eyed vireo; SCTA, scarlet tanager; WEWA, worm-eating warbler; WOTH, wood thrush. Adapted from Robinson* et al. *(1995).*

been shown to be higher near edges. Cowbirds are also more abundant near clearcut edges in some landscapes (Thompson *et al.* 1992), but not in others (Winslow *et al.*, in press). Mating success of ovenbirds is also lower near edges (Gibbs and Faaborg 1990, Van Horn *et al.* 1995). Even trails may attract nest predators and parasites (Hickman 1990). Other studies that showed only weak or no edge effects were conducted in landscapes where nesting success is low everywhere and there may be no true habitat interiors (Robinson and Wilcove 1994; Thompson *et al.*, in press). In such habitats, cowbird and nest predator populations appear to saturate the available forests (Marini *et al.* 1995, Heske 1995). Even in severely fragmented forests, however, some edges (e.g., powerline corridors, campgrounds) are associated with increased levels of nest predation and parasitism (Robinson, unpubl. data). A particularly promising line of investigation will be to compare edge effects in different kinds of landscapes. Effects of edges created by logging, for example, may be most apparent in primarily forested rather than in mostly agricultural landscapes (Thompson *et al.* 1992; Winslow *et al.*, in press). Studies of the effects of different kinds of edges on birds would also be useful to managers (Ratti and Reese 1988). While edge-related declines in reproductive success have

been reported, no field studies have demonstrated their impact on populations (though they have simulated, Temple and Cary 1988, Thompson 1993).

Habitat heterogeneity.—The pronounced area-sensitivity of many NTMBs and other edge-related problems with nesting success have led some researchers to question the value of managing to promote habitat heterogeneity at the expense of large, contiguous tracts (Faaborg 1980, Reese and Ratti 1988, Robinson 1988). The kinds of management practices used to promote populations of some game animals may have detrimental effects on habitat specialists. Heterogeneity is needed at some (large) spatial scale, however, to meet the diverse habitat needs of NTMBs. Landscapes that consist of a mosaic of small (<5-ha) habitat patches are often dominated by habitat generalists and may lack habitat-interior specialists. A productive area for future research would be to develop management practices for areas managed for hunting that increase the value of these areas for NTMBs, without negatively affecting game populations (or perhaps even increasing them).

An important consideration for managers is that NTMBs have diverse habitat requirements. Even within a tract, what is productive habitat for one NTMB may be unproductive for another. Kentucky warblers for example appear to nest most successfully in older forests along streamsides and least successfully in evenage clearcuts and tree plantations (Morse 1996). Worm-eating warblers, on the other hand, nest equally successfully in older (>15-year) clearcuts as they do in older (>80-year) forests (Robinson, unpubl. data). Both species, however, respond well to selective logging (Robinson and Robinson, unpubl. data). On a regional spatial scale, managers need to ensure that a mixture of forest types and successional stages are available to maintain habitats for all NTMBs typical of that region. In mostly forested landscapes, this regional heterogeneity can be maintained by a mix of silvicultural practices (Thompson *et al.* 1996). In mostly agricultural landscapes where forest cover is limited, management practices should emphasize maintaining and enhancing forest cover rather than maximizing local habitat heterogeneity. In these landscapes, selective logging may be preferable to even-aged forest management (Thompson *et al.* 1995).

Metapopulation dynamics.—The "source-sink" metaphor (Pulliam 1988) for population dynamics appears particularly applicable to midwestern landscapes. Small, isolated woodlots in primarily agricultural landscapes appear to be ecological "traps" (sensu Gates and Gysel 1978) that attract birds, but fail to provide the conditions necessary for successful nesting as a result of high levels of parasitism and predation (Robinson 1992; Trine *et al.*, in press; Donovan *et al.* 1995a,b). Populations in these habitats may act as drains on regional populations and are maintained by immigrants from elsewhere (the "rescue effect" of Brown and Kodric-Brown 1977). A potentially disturbing aspect of the rescue effect is that populations in woodlots with extremely low nesting success can appear to be stable over the long term (Brawn and Robinson 1996). Problems with inferring local population health may be particularly severe for NTMBs in which young rarely return to breed in their natal area. This uncoupling of local reproductive success and population dynamics may be unique to fragmented landscapes. Holmes *et al.* (1988) and Sherry and Holmes (1992) found a strong correlation between reproductive success and population changes in subsequent years in the unfragmented forests of northern New England. Sink habitat, however, may not destabilize regional populations as long as adequate source habitat remains (Howe *et al.* 1991).

The results of Robinson *et al.* (1995a) suggest that a few key large forest tracts may be sustaining forest bird populations in most of the Midwest. Although we are a long way from being able to prove that this is the case, a prudent conservation strategy would be to preserve these large forest tracts. Because national forests form the core of most large tracts in the Midwest (see Thompson *et al.* 1996), any change in ownership could have a detrimental effect on regional populations if it results in increasing agricultural, residential, and recreational development.

Promising areas for future research include determining how much parasitism and predation key NTMBs can tolerate before source populations become sinks (the source-sink threshold of Pulliam 1988). Species may vary greatly in their vulnerability to cowbird parasitism (e.g., May and Robinson 1985). Donovan *et al.* (1996) provide examples of the kinds of data needed for source/sink analyses

(see also Donovan1995). As we obtain better demographic data, we will also be able to improve our estimates of the amount of source habitat necessary to balance sink habitats.

HABITAT LOSS

Most of the chapters in this book deal with the consequences of the loss of various key midwestern habitats. Virtually all native habitats in the Midwest have undergone drastic reductions since pre-settlement times. In the following section, I briefly review the key threats posed by habitat loss and refer the readers to the more detailed discussions in the following chapters.

Loss of floodplain forest has been particularly severe in the Midwest (Pashley and Barrow 1993, Knutson *et al.* 1996). To help reverse this trend, the Nature Conservancy, U.S. Fish and Wildlife Service, and Illinois Department of Natural Resources have formed a coalition to restore 60,000 acres of the Cache River wetlands in southern Illinois. Such projects may be necessary if we are to regain functioning floodplain forests of sufficient size to contain area-sensitive species and those that require natural disturbances (see below). Floodplain forest is especially important for Swainson's warbler (Eddleman 1978, Eddleman *et al.* 1980), Mississippi kites (Evans 1981), cerulean warblers (Robbins *et al.* 1992, Vanderah 1995), and red-shouldered hawks (Bednarz and Dinsmore 1981, Hands *et al.* 1989a), all of which are area-sensitive and at least locally endangered and a high regional priority (Thompson *et al.* 1993).

Central Hardwoods forests form the largest remaining tracts of native wildlife habitat remaining in the lower Midwest (Thompson *et al.* 1996). Their management is particularly crucial because of their economic importance. The Missouri Ozark Forest Ecosystem Project (Kurzejeski *et al.* 1994) promises to provide major insights into the effects of silvicultural practices on NTMBs (see also Thompson *et al.* 1995). Fragmentation by agricultural development is already known to pose a severe threat to NTMBs (Robinson *et al.* 1995a). Determining the extent to which silvicultural practices fragment the Central Hardwoods forest or provides needed habitat diversity remains a top research priority (Thompson 1993, Thompson *et al.* 1996, Annand and Thompson 1997). Because large, unfragmented tracts of Central Hardwoods forest already exist, the major threat to this ecosystem would be from changes in land ownership or management that might cause fragmentation of currently contiguous tracts.

Savanna/burned forests and barrens have also been reduced by a century of strict control of fires (Thompson *et al.* 1996). We know little about the dependence of NTMBs on burning except to say that many bird species of open, savanna-like habitats have also adapted well to human-modified habitats (Rotenberry *et al.* 1993, 1995; Robinson 1994; Brawn 1994). The Bachman's sparrow (not a NTMB) is perhaps the most notable exception because it appears to require large barrens and glades (Chambers 1994).

Grasslands have undergone perhaps the most severe reduction of any midwestern habitat (Herkert 1991, 1994a,b; Samson 1980; Johnson 1996; Herkert 1996). In addition, many surrogate grasslands such as hayfields and CRP fields are now being managed in such a way that successful nesting is often impossible (Bollinger and Gavin 1992; Herkert 1994a,b). Given their sensitivity to both area (Samson 1980; Herkert 1994a,b) and management practices, it is not surprising that grassland birds have declined dramatically (reviewed in Herkert 1991). Reversing these population declines is a top priority for the Midwest (even though many species are not NTMBs). Major restoration efforts such as the proposed Midewin Tallgrass Prairie near Chicago may be necessary to help grassland bird populations.

Northern Hardwood/Conifer Forests still exist in large tracts in the upper Midwest (Howe *et al.* 1996). As with Central Hardwoods ecosystems, the primary potential threats come from silvicultural practices and any changes in land ownership that might open up large, contiguous tracts to agricultural and residential development. The shift from late successional (spruce) to early successional (aspen-dominated) plant communities created by silvicultural activities likely has had a major impact on bird communities as well.

Agricultural landscapes formerly provided rich habitats for many species of NTMBs (Graber and Graber 1963), but are now being managed so intensively for row crops that they have little value for birds (Paruk 1990; Warner

1994; Rodenhouse *et al.* 1993, 1995; Koford and Best 1996). Creative use of government subsidies (e.g., using them to make larger habitat patches and delaying hay harvesting) may offer the best short-term hope for improving agricultural landscapes for NTMBs (Rodenhouse *et al.* 1992, Freemark *et al.* 1995). Agricultural practices may also expose birds to potentially hazardous pesticides and other environmental contaminants (Gard and Hooper 1995).

HABITAT STRUCTURE AND COMPOSITION

In addition to outright habitat loss described in the previous section, changes in habitat structure and composition pose significant threats to NTMBs. Below, I review some of the ways that changes in vegetation structure and composition can threaten populations of NTMBs.

Loss of oaks.—In the absence of fire, it is very likely that the relative importance of oaks will decline as they are replaced by late-successional, shade-tolerant species (Abrams 1992; Thompson *et al.* 1996). Although there are few data on use of particular tree species by NTMBs, a loss of oaks may negatively affect cerulean warblers (Robbins *et al.* 1992, Vanderah 1985) and result in a general deterioration of foraging conditions for some spring migrants (Graber and Graber 1983, Moore *et al.* 1993). Managing some areas to retain oak dominance (especially white oaks) seems like a prudent conservation strategy. Loss of oaks killed during impending gypsy moth invasions may add to the problems faced by oak-dependent wildlife. A productive area for future research is the potential role of birds in controlling levels of folivorous insects that might otherwise decrease forest health and change forest composition (Marquis and Whelan 1994, Whelan *et al.* 1989). Unfortunately, birds appear to have little impact on gypsy moth outbreaks (Smith 1985).

Loss of other tree species.—At least a few species appear to select nesting habitats based on particular tree species (Kahl *et al.* 1985, Mossman and Lange 1982, Robbins *et al.* 1992, Vanderah 1995). Continued loss of sycamores as a result of disease, for example, may negatively affect cerulean warblers (Vanderah 1985), northern parulas and yellow-throated warblers (Robinson, unpubl. data). Losses of dogwoods to anthracnose disease may reduce nesting substrate for species such as the wood thrush that routinely use them for nesting (S. Robinson, unpubl. data), and other species use their fruit during fall migration. More data are needed before we can make recommendations about how to manage tree species composition (Holmes and Schultz 1988).

Tree monocultures.—The replacement of diverse, mixed-species forests with planted monocultures (e.g., pines, walnuts, and sweetgum) has unpredictable effects on bird communities. Pine monocultures in Illinois and Indiana have greatly increased populations of pine and yellow-throated warblers in the state and provided nesting sites for Cooper's and sharp-shinned hawks in Missouri (Ehrlich and Drickamer 1993; Robinson, unpubl. data; Kritz 1989; D. Whitehead, pers. comm.). Pine plantations also contain significant populations of veeries, ovenbirds, hooded and Kentucky warblers, and summer tanagers in Illinois (Robinson, unpubl. data). White pine plantations in northern Illinois and Indiana have also been colonized by nesting populations of black-throated green warblers and brown creepers (Robinson, unpubl. data; D. Whitehead, pers. comm.). Pine plantations therefore appear to provide a surrogate habitat for many NTMBs. Cerulean warblers sometimes nest in black locust monocultures (Vanderah and Robinson, unpubl. data). There are few data, however, on nesting success of birds in plantations; Kentucky warblers suffer high parasitism and predation levels in a sweetgum plantation in southern Illinois (S. Morse, unpubl. data). Furthermore, it is possible that non-native plant communities may be ecological "traps" for extralimital breeders that ordinarily would nest further north or south. The use of plantations by NTMBs is clearly worth further study, especially in industrial forests.

Loss of understory.—Excess deer populations and (potentially) burning can result in a substantial reduction in the shrub layer (Thompson *et al.* 1996; Alverson *et al.* 1988). Species that require dense shrub layers such as Kentucky warblers (Wenny *et al.* 1993) may suffer increased nest predation as a result of reduced nest cover. Species preferring more open understories may benefit as a result (e.g., ovenbird: Wenny *et al.* 1993, Kahl *et al.* 1985).

Changes in plant species composition resulting from overbrowsing also may have longer-term negative consequences for canopy species. Deer management therefore may be critical to the long-term maintenance of NTMB populations.

Floodplain disturbance.—The scarcity of unconstrained, naturally meandering rivers in the Midwest may be creating long-term problems for several species. Natural disturbances along floodplains may have created the original habitat of many species (Zimmerman and Tatschl 1975, Rotenberry *et al.* 1993, Pashley and Barrow 1993), including cane-dependent Swainson's warblers (Eddleman *et al.* 1980) and Bachman's warblers (Remsen 1986, Hamel 1986). Cerulean warblers nest abundantly along natural levees in southern Illinois (Vanderah 1995) and prothonotary warblers show a strong preference for backwater habitats (J.P. Hoover, unpubl. data; Knutson *et al.* 1996). Red-shouldered hawks (Bednarz and Dinsmore 1981) and Mississippi kites (Evans 1981) also depend partly on openings in forest canopies created by large-scale disturbances. Managing for these species may require restoring natural floodplain dynamics (a difficult task) or mimicking natural disturbances (Rotenberry *et al.* 1993, 1995).

Shrubland/edge species.—Although many species that depend upon shrubland/edge habitats have benefited from human settlement of the Midwest (Johnston 1947, Karr 1968) some are now showing steep population declines (Askins 1993). Several species of special concern (listed in Thompson *et al.* 1993) in the Midwest (e.g., Bell's vireo: Hands *et al.* 1989b, golden-winged warbler: Hands *et al.* 1989b) depend on early successional habitats that are currently being created almost entirely by human land uses. Private, industrial forests and areas managed for game species may provide opportunities to manage for birds of early successional habitats that also require large tracts (e.g., prairie warbler: Nolan 1978, Annand and Thompson 1997). There also is a critical need for data on the effects of different *kinds* of edges on birds. Avoiding abrupt edges may increase nesting success of both forest birds (Ratti and Reese 1988) and edge-nesting birds (A. Suarez, K. Pfennig, and S.K. Robinson, submitted ms.). Similarly, natural, complex edges may provide better nesting habitat for some forest birds (e.g., Acadian flycatcher: L. Chapa and S.K. Robinson, unpubl. data).

Loss of old-growth forest.—Although the Midwest appears to lack an old-growth specialist such as the spotted owl, the cerulean warbler may depend in part on old-growth forests (Vanderah 1995). Pileated woodpeckers may also depend upon large trees (S. Robinson, pers. obs.) and the hooded warblers use treefall gaps in old-growth forests. A prudent conservation strategy in any managed landscape would be to maintain core areas of old-growth timber, especially in areas currently occupied by cerulean warblers. Natural disturbances characteristic of old-growth may also provide optimal nesting habitat for gap-dependent species (Noss 1991; Robinson, unpubl. data).

Exotic plant invasion.—The rapid encroachment of plants such as Japanese honeysuckle, buckthorn, and garlic mustard have as-yet undetermined effects on NTMBs. Certainly, many birds use these plant species for nest sites, but we do not yet know if nesting success is as high in non-native vegetation, or if foraging opportunities are as good.

Snag loss.—Any management activity that would reduce the density of snags could affect some NTMBs adversely, although only a few NTMBs nest in snags or use them as song perches. Snag management is made even more complicated because they are used frequently by cowbirds as perches, possibly to search for nests (Robinson *et al.* 1995b).

Mid-successional dynamics.—We know little about how bird community composition changes through the middle stages of succession. Even-aged forests of the White Mountains of New Hampshire appear to undergo a major peak of diversity and abundance in the 60-90 year range post-cutting (Holmes *et al.* 1986; R.T. Holmes, pers. comm.). If this proves to be a general result, then we may need a more complex mosaic of successional stages than previously assumed.

OTHER THREATS

Global climate change.—Climate change could have far-reaching consequences for midwestern birds, but we are not yet in a position to predict what they may be (Rotenberry *et al.* 1993, 1995; Root and Schneider 1995). The flexibility and broad geographic ranges of most breeding NTMBs, however, indicate that effects on breeding populations may be subtle.

Loss of stopover habitat.—The isolated woodlots of the agricultural Midwest and the coastal woodlots of the Great Lakes may be critical habitat for NTMBs during migration (Blake 1986; Moore and Simons 1992; Winker *et al.* 1992; Moore *et al.* 1993, 1995; Ewert and Hamas 1996). Maintenance of these habitat patches may be critical to more northerly breeding species (Moore *et al.* 1993). Acquisition of shoreline property on the Great Lakes is especially timely (D. Ewert, pers. comm.). Plant species composition of stopover sites is particularly likely to be a crucial management consideration (Graber and Graber 1983, Moore *et al.* 1993). Some stopover sites may fail to provide enough food for migrants to gain mass (Winker *et al.* 1992).

Loss of winter habitat.—Over the long run, some species may decline in the Midwest regardless of how well we manage our ecosystems (Terborgh 1989, Robbins *et al.* 1992, Petit *et al.* 1993, Faaborg *et al.* 1996). At present, however, we know too little about population regulation in any NTMBs to make the assumption that there is nothing we can do in the Midwest. Even the cerulean warbler, which is severely threatened on its eastern Andean wintering grounds (Robbins *et al.* 1992) is also threatened by forest fragmentation and cowbird parasitism on the breeding grounds (Robbins *et al.* 1992, Vanderah 1995).

Acid rain.—The extremely severe local population declines documented by the BBS may be at least partly caused by loss of spruce forests as a result of acid rain (James *et al.* 1996, James and McCulloch 1995). This problem is likely to be much less severe in the Midwest than in the East.

Food supply.—Conceivably, use of pesticides and fragmentation may be disrupting food supplies for many insectivorous birds (Gard *et al.* 1993, Gard and Hooper 1995). Preliminary data from Illinois fragments and other midwestern forests suggest that most nests that escape predation fledge most of the young that hatch, which indicates that food supplies may rarely be limiting (S. Robinson, unpubl. data; D. Whitehead, pers. comm.; J. Faaborg, pers. comm.). In contrast, in extensive forests in the Northeast, many species may be limited by food and some may even depend upon pulses of high productivity during outbreaks of defoliating caterpillars (Holmes *et al.* 1992). More studies over a greater range of geographical areas and climatic extremes would help determine the extent to which food supplies may be limiting.

Shiny Cowbirds.—Although it has not yet become abundant in the Midwest, the shiny cowbird may pose a new threat to many species (Robinson *et al.* 1995b). Shiny cowbirds regularly parasitize cavity-nesters, which could greatly expand the number of host species parasitized intensively.

Extreme Flooding.—The unprecedented 1993 midwestern floods killed many trees in the Mississippi floodplain but we know little about the effects on the bird communities. Additional flood events may further damage existing forests, but could also increase pressure to convert marginal farmland to wetland habitat that would be used extensively by NTMBs and other wildlife.

Drought.—There are several studies showing effects of drought on populations (e.g., Blake *et al.* 1992), productivity (DeSante *et al.* 1993), and community structure (Smith 1982). The drought of 1988 was associated with low population levels of forest birds in central Illinois (Robinson 1992), but we have no data on productivity during this period. Populations have largely recovered since the drought ended (Brawn and Robinson 1996), which suggests that effects of drought may be short-lived and therefore may not pose a long-term threat to NTMBs.

TRACT SIZE AND THE CONSERVATION OF MIGRANTS

Other papers in this symposium provide ecosystem-specific guidelines for conservation efforts in midwestern landscapes. One general approach to NTMB conservation that solves many, if not most of the problems identified in this review is to enlarge tract sizes and conserve existing large tracts (table 2). Very large habitat blocks (>25,000 acres) can meet the needs of all area-sensitive birds and are large enough to incorporate natural disturbances (Rotenberry *et al.* 1993). The intensive agricultural development in Illinois limits opportunities to conserve or restore tracts of this size, but several sites are reasonably large including the Cache River Bioreserve/Wetlands (60,000 acres) and the Prairie Parklands/Joliet Arsenal/Midewin Tallgrass Prairie site (43,500

Table 2.—*Threats reduced by emphasizing the management of large tracts ("macrosites").*

Threat	How the threat is reduced in large tracts
Cowbird parasitism	Fewer feeding opportunities, longer commuting distances, fewer edges
Nest predation	Reduced populations of some species subsidized by agricultural waste, fewer edges, more top predators to control medium-sized predators
Area sensitivity	Large enough to include all species, including raptors
Lack of natural disturbance	Large enough to manage for several kinds of disturbance simultaneously, possibly restore floodplain activities
Edge effects	Fewer edges, especially abrupt agricultural ones
Habitat heterogeneity	Opportunity to manage several large, contiguous habitat patches without too much local heterogeneity
Metapopulation collapse	Opportunity to create "source" habitat to recolonize nearby smaller tracts
Habitat loss	Opportunity to restore floodplain (e.g., Cache River Wetlands) and grassland (e.g., Prairie Parklands site)
Deer browse	Reduction of deer populations by natural predation, herd management, and distance from agricultural areas
Tree species composition	Opportunity to restore large areas of keystone plant species (e.g., white oaks)
Loss of old-growth	Opportunity to set aside "core" areas of old growth
Food supply	Isolation of core area from pesticides and other agrochemicals

acres). States such as Missouri, Indiana, Wisconsin, and Minnesota have the potential to preserve and restore much larger forest tracts.

How large tracts have to be to benefit NTMBs remains an open question. Robinson *et al.* (1995a) found substantial reductions in parasitism and predation levels in landscapes with an average tract size of 10,000-25,000 ha. Because forest cover covaried so closely with tract size, however, Robinson *et al.'s* (1995a) data cannot be used to identify tract sizes necessary to reduce parasitism and predation levels. Nor do we yet have firm targets for what levels of parasitism and predation are acceptable because of interspecific differences in tolerance of parasitism and predation (May and Robinson 1985, Martin 1992). As a general rule, daily nest predation rates of 4 percent or less and parasitism levels of 25 percent or less should give most NTMBs at least a chance of having self-sustaining populations (May and Robinson 1985; Donovan *et al.* 1995a,b; Trine *et al.*, in press). Clearly, however, we need better demographic data before we can set targets for minimum tract sizes. It is likely, however, that conservation strategies will have to be tailored to specific landscapes. What works best in agricultural landscapes may not be the best approach in mostly forested landscapes (Robinson *et al.* 1995). Enlarging tract size may be most crucial in agricultural landscapes. In contrast, managers in mostly forested landscapes should be

concerned primarily with maintaining a representative mix of forest types to meet the habitat requirements of all forest species native to the region.

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A MULTI-SCALE ASSESSMENT OF THE GEOGRAPHIC AND ECOLOGICAL DISTRIBUTION OF MIDWESTERN NEOTROPICAL MIGRATORY BIRDS

John R. Probst[1] and Frank R. Thompson, III[2]

ABSTRACT.—Multi-scale assessments of species status are valuable for natural resource management because of interactions between local and regional populations. Such assessments can be broad and cost-effective if done by step-wise successive approximation. We described the distribution of landcover and neotropical migratory birds (NTMBs) in the Midwest at several different geographic and ecological scales. First, we mapped the distribution of major land-cover types in the Midwest. Next, we identified 187 NTMBs that breed in the Midwest, and 47 regional high-priority species for conservation from the Partners in Flight (PIF) NTMB database. We also identified 57 Midwestern species that are declining nationally based on the Breeding Bird Survey (BBS). We report the number of species, number of priority species, and number of declining species across seven ecological provinces within the region. We reviewed literature to determine important breeding habitats and report the number of species, number of priority species, and number of declining species by general land-cover types and by finer-resolution habitat types. At the next level of resolution we suggest ordinating species along relevant ecological gradients, and present two examples. This type of multi-scale assessment provides information on species at different levels of current concern, and identifies ecological provinces, landcovers, and habitats with large numbers species, priority species, and declining species.

INTRODUCTION

Both scientists and the public have become disillusioned with the narrow, expensive, crisis management of Endangered Species. Species viability cannot be insured only by evaluating and improving local habitats (e.g., USFWS 1980) without of a general conservation plan covering much (e.g., Thomas *et al.* 1990) or all of the species range (e.g., Probst and Weinrich 1993) . To supplement single-species approaches, biologists have been developing more holistic, multi-species approaches to conservation (e.g., Scott *et al.* 1993). Such approaches should not only include most vertebrate species, but should also be integrated with disturbance regimes and forest harvesting (e.g., Probst and Crow 1991, Thompson *et al.* 1993). Because of the interaction between local and regional populations (e.g., Askins *et al.* 1987, Probst and Weinrich 1993), multi-scale assessments of species (or other resources) are useful in addressing both process and cumulative effects, and they can be both broad and cost-effective if done by step-wise, successive approximation of population processes (Freemark *et al.* 1993). By placing local decisions in a regional, multi-resource context, land managers and planners can direct local planning to meet different but complementary objectives.

This symposium (Thompson 1996) presents a multi-scale approach to the conservation of neotropical migratory birds (NTMBs) in Midwestern North America. Many of the papers in this symposium review landscape and local factors affecting the status of NTMBs (Johnson 1996, Herkert *et al.* 1996, Howe *et al.* 1996, Knutson *et al.* 1996, Koford and Best 1996, Thompson *et al.* 1996). We provide context for

[1] *North Central Forest Experiment Station, 5985 Hwy K, Rhinelander, WI 54501-0898.*

[2] *North Central Forest Experiment Station, 1-26 Agriculture Bldg., Columbia, MO 65211.*

these papers with a coarser-grain, successive-approximation overview at broader geographic and ecological scales to facilitate subregional assessment and management. We began by mapping the distribution of major land-cover types in the Midwest. Next we identified Midwestern NTMBs from the PIF NTMB database and regional high-priority species for conservation based on this database. We also identified Midwestern species that are declining nationally based on the Breeding Bird Survey. We determined the distribution of NTMBs across 7 ecological provinces within the region. We then examined the distribution NTMBs, high-priority NTMBs, and declining NTMBs by general land-cover types, by finer-resolution habitat types, and by landcover types within ecological provinces. And finally, as an example of a finer level of resolution, we hypothesize habitat relationships by ordinating species along two ecological gradients.

METHODS

Land Cover in Midwestern U.S.

We assessed landcover in the region from AVHRR imagery (1990 Conterminous U.S. Land Cover Characteristics Data Set CD-ROM, EROS Data Center, USGS, Sioux Falls, South Dakota). These data are 1-km resolution and were developed for large-scale assessments. We extracted coverage for all or much of 15 Midwestern States and pooled vegetation classes into 11 land-cover classes (reported in results). Because of the low resolution of these data, landcover classes contain mixed vegetation types. Therefore, these data should be used only for assessing large-scale patterns.

Midwestern NTMBs

We used the PIF NTMB database to identify Midwestern NTMBs and to determine the distribution and status of species within the region. This database includes long- and short-distance NTMBs (Gauthreaux 1992), listed by State and physiographic regions. We defined Midwestern species as those occurring in any of 24 physiographic regions in midwestern North America (fig. 1). These physiographic regions were delineated for the Breeding Bird Survey and are used by the PIF database. This definition of Midwestern North America was somewhat problematic because some of the boreal, grassland, and forest regions extend across the continent and include eastern and western avifaunas. However, we thought this approach was superior to one based on political (State) boundaries and it is compatible with ongoing PIF conservation efforts.

The PIF NTMB database also includes information used to prioritize species for conservation efforts. Species priority scores are based on 7 criteria: global abundance, breeding distribution, winter distribution, threats on breeding grounds, threats on wintering grounds, importance of the area under consideration (State or physiographic region) to the species, and population trend (Hunter *et al.* 1993, Carter and Barker 1993). Each criterion is scored from 1 to 5, and species are prioritized by their total score (35 = highest priority). We identified regional priorities by calculating a regional priority score for each species from these data. For each species, we calculated the mean value for each of the 7 criteria across physiographic regions, except for the importance of area criterion. Our regional assessment of this criterion needed to take into account the total value of all physiographic regions (not their mean). We transformed scores for this criterion for each species in each physiographic region back to an estimated percentage of the species range. To do this we assumed the percentage was the midpoint of the interval used to assign the original score (i.e., a score of 3 indicated 11 to 25 percent of the species range was in the region, so we assigned the midpoint, 18 percent). We calculated the sum of these percentages and then re-scored this regional percentage 1-5 based on the original criteria. We then summed scores for all seven criteria, now all adjusted to reflect regional values, to create our regional priority score. As with the original physiographic scores, these scores could range from 7 to 35. Various criteria have been used to select priority species from these scores (Carter and Barker 1993, Thompson *et al.* 1993). In this paper, we refer to species with priority scores greater than the 75th percentile (22.2) as priority species.

We also identified midwestern NTMBs that had declining populations. We examined population trends of the 187 midwestern NTMBs for the United States calculated from the Breeding Bird Survey (BBS) and identified species with significant population declines ($P < 0.1$) for the period 1966-1994 (information provided by Bruce G. Peterjohn, National Biological Service, Patuxent Environmental Science Center, Laurel, Maryland).

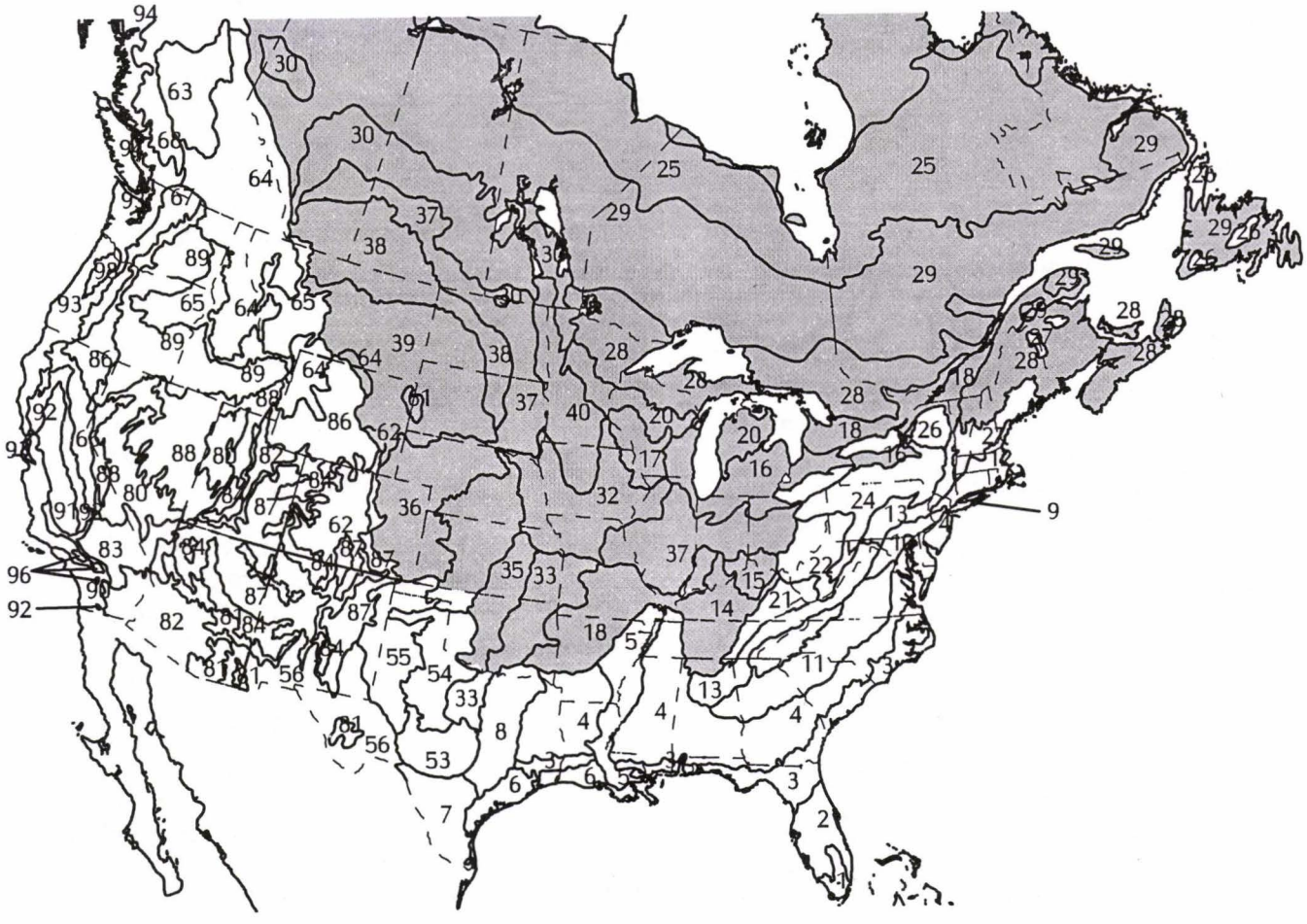


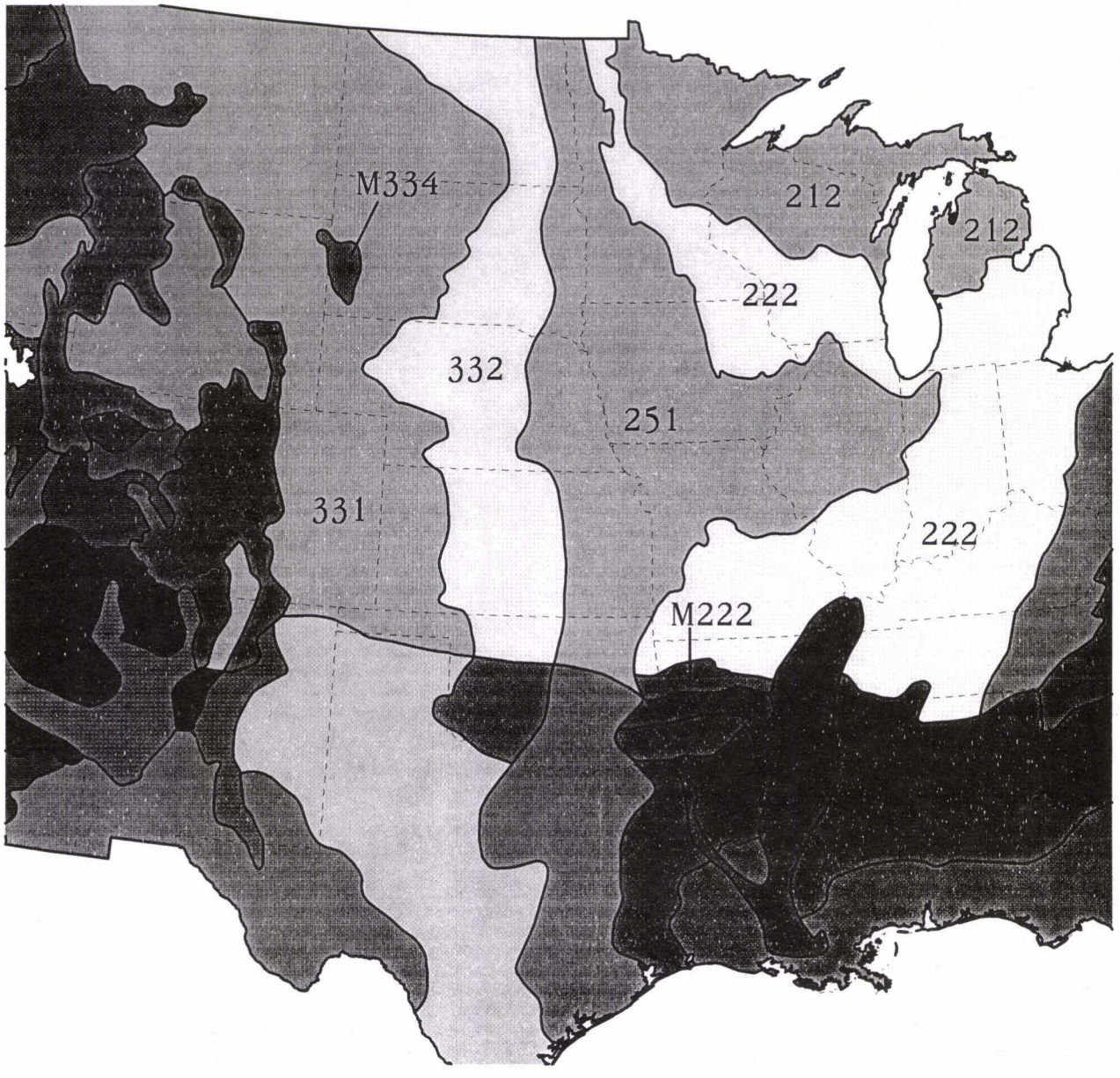
Figure 1.—*Physiographic regions of the United States used by the Breeding Bird Survey and the Partners in Flight Database. We summarized data from the Partners in Flight Database for the shaded regions in this map to identify NTMB species and priority species in the Midwest.*

Distribution of NTMBs Across Ecological Provinces

We determined the occurrence of midwestern NTMBs and priority NTMBs across seven ecological provinces in the Midwest (fig. 2). Ecological provinces are part of the National Hierarchical Framework of Ecological Units adopted by the USDA Forest Service (Bailey *et al.* 1994, McNab and Avers 1994). At this scale patterns of species distribution can be examined at geographically broad but ecologically defined land units. We identified species associated with ecological provinces by relating them to the physiographic region used in the original database. Species were included in a province from all physiographic regions that overlapped a province by approximately 15 percent or more.

NTMB Distribution Across Land Cover and Habitats

We identified up to five breeding habitats used by each species based on a literature review of published midwestern studies (F. Thompson and J. Probst, on file) and our own experience. Habitat types were: agriculture (cropland, pasture, fence rows, farmyards), developed (suburban, urban, commercial development), grassland (prairie, rangeland), shrub-sapling (oldfields, regenerating forest), shrub wetlands, upland conifer forest, lowland conifer forest, upland deciduous forest, lowland deciduous forest, savanna, and specialized (nesting requirements such as buildings or eaves, cliffs or banks). For initial, coarser-grained assessments, we used a reduced land cover list:



PROVINCES	
212	Laurentian Mixed Forest Province
222	Eastern Broadleaf Forest (Continental) Province
M222	Ozark Broadleaf Forest - Meadow Province
251	Prairie Parkland (Temperate) Province
331	Great Plains - Palouse Dry Steppe Province
332	Great Plains Steppe Province
M334	Black Hills Coniferous Forest Province

Figure 2.—*Ecological Provinces of the Midwestern United States; Provinces are one level in the Forest Service National Hierarchical Framework of Ecological Units (adapted from Bailey* et al. *1994 and McNab and Avers 1994).*

aquatic, agriculture/developed (agriculture + developed + specialized), grassland, shrub/sapling (shrub-sapling + shrub wetlands), forest (upland conifer forest + lowland conifer forest + upland deciduous forest + lowland deciduous forest) and savanna. We also determined the distribution of species by landcovers within ecological provinces. For a finer scale assessment we also report numbers of species by habitats. Species were often associated with more than one landcover and habitat and counted in more than one category.

Arranging species on ecological gradients is a more general method for describing habitat use. Gradients can be used as a basis for multiple characterizations of species associations, including more common classifications systems. As an example, we hypothesized relationships of forest birds along gradients of seral stages and coniferous to deciduous trees. These hypotheses can guide verification through surveys, including modification of gradient relationships across geographic ranges.

Geographic Links and Conservation Planning

Many Midwestern species' ranges extend beyond midwestern North America so effective conservation may require coordination of conservation efforts across geographic areas. To demonstrate this, we identified Midwestern high priority species that had a large portion of their range outside midwestern North America. We examined range maps of these species and noted if a significant portion of their range and ecosystem was in northeast, southeast, southwest, or northwest North America.

RESULTS AND DISCUSSION

Land Cover in Midwestern U.S.

Dominant landcover in the region is cropland, woodland cropland mix, forestland, and grassland (fig. 3). Savanna, desert shrubland, shrubland/grassland, and grassland/cropland make up smaller proportions of the area. There are strong regional patterns in landcover in the Midwest, with heavily forested landscapes in the northern and southern regions, grasslands in the western region, and predominately cropland and fragmented forest and grasslands in the central region. Johnson (1996), Herkert *et al.* (1996), Howe *et al.* (1996), Knutson *et al.* (1996), Koford and Best (1996), and Thompson *et al.* (1996) provide more detailed information on the distribution of some Midwestern ecosystems.

Midwestern NTMBs and Their Distribution

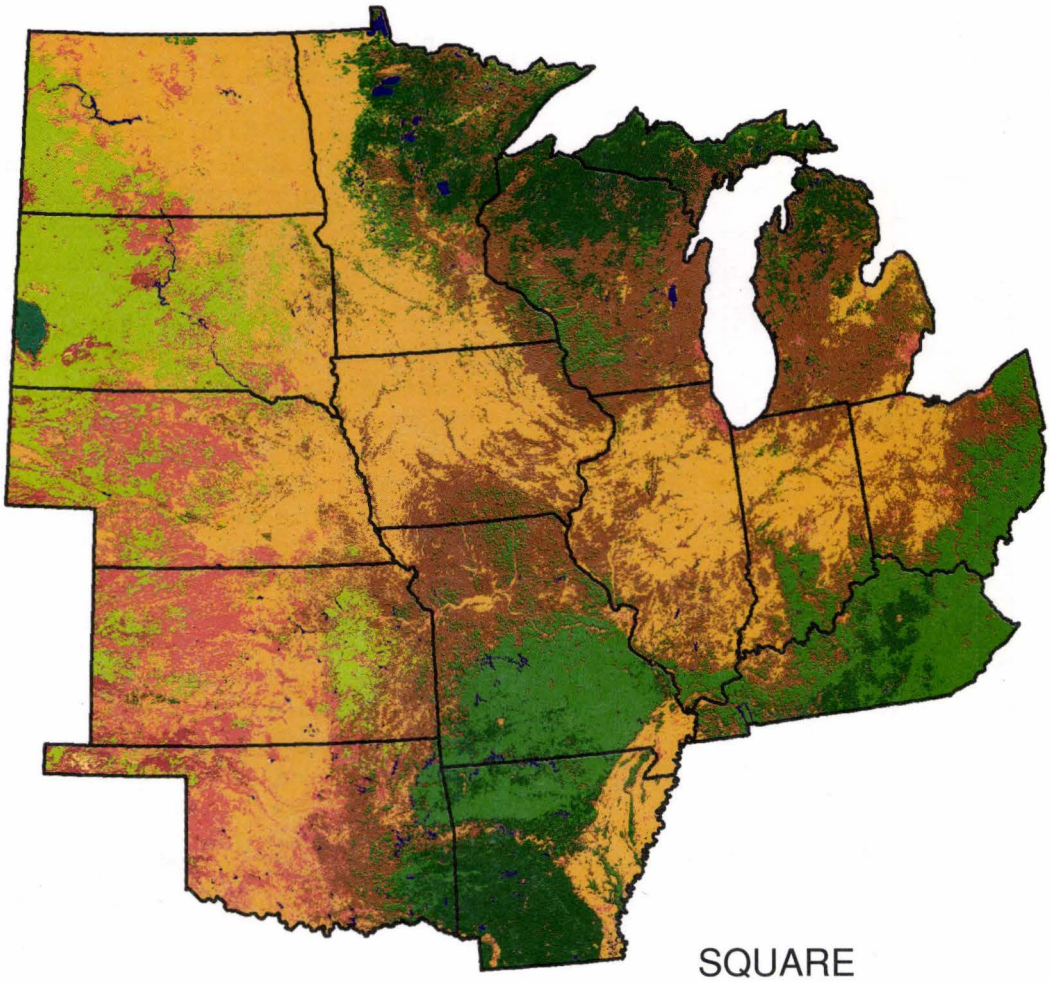
We identified 187 species that occurred within the region (Appendix 1). Forty-seven species had priority scores greater than the 75th percentile (22.2). These priority species represent diverse taxonomic orders of birds and use a wide range of habitats. Fifty-seven of the 187 Midwestern NTMBs were declining in the US (Appendix 1).

Distribution of NTMBs Across Ecological Provinces

The number of species within Ecological Provinces ranges from 81 in the Black Hills Coniferous to 136 in Prairie Parkland. Number of priority species ranges from 10 in the Black Hills Coniferous to 33 in the Prairie Parklands. Most of the midwestern provinces contain high numbers of NTMB species and priority species because these provinces represent both east-west and north-south continental ecotones, and contain prairies, forest and wetlands. Trends and species numbers appear related to the geographic scope and habitat diversity of the Provinces. For example, the Prairie Parkland, with high numbers of species and priority species, was originally a forest-openland mosaic that has largely been converted to agriculture.

NTMB Distribution Across Land Cover and Habitats

The 187 species of NTMBs were broadly distributed across land cover types. Approximately 51 percent of these species were associated with shrub/sapling land cover, 50 percent with forest, 25 percent agricultural/developed land cover, 24 percent with grassland, 21 percent with savanna, and 4 percent with aquatic landcover (percentages sum to >100 percent because species were associated with >1 habitat) (fig. 4). The distribution of priority species shifted slightly from agricultural/developed habitats to grassland land cover; 51 percent were associated with forest, 47 percent with shrub sapling, 34 percent with grassland,



COVER TYPE	SQUARE KILOMETERS
CROPLAND	805172
GRASSLAND/CROPLAND	184148
WOODLAND/CROPLAND	536990
GRASSLAND	232442
DESERT SHRUBLAND	1900
SHRUBLAND/GRASSLAND	26643
SAVANNA	32937
DECIDUOUS FORESTLAND	327851
CONIFEROUS FORESTLAND	9227
MIXED FORESTLAND	223473
WATER	24269

Figure 3.—Land cover of the Midwestern United States interpreted form AVHRR imagery: see text for methods.

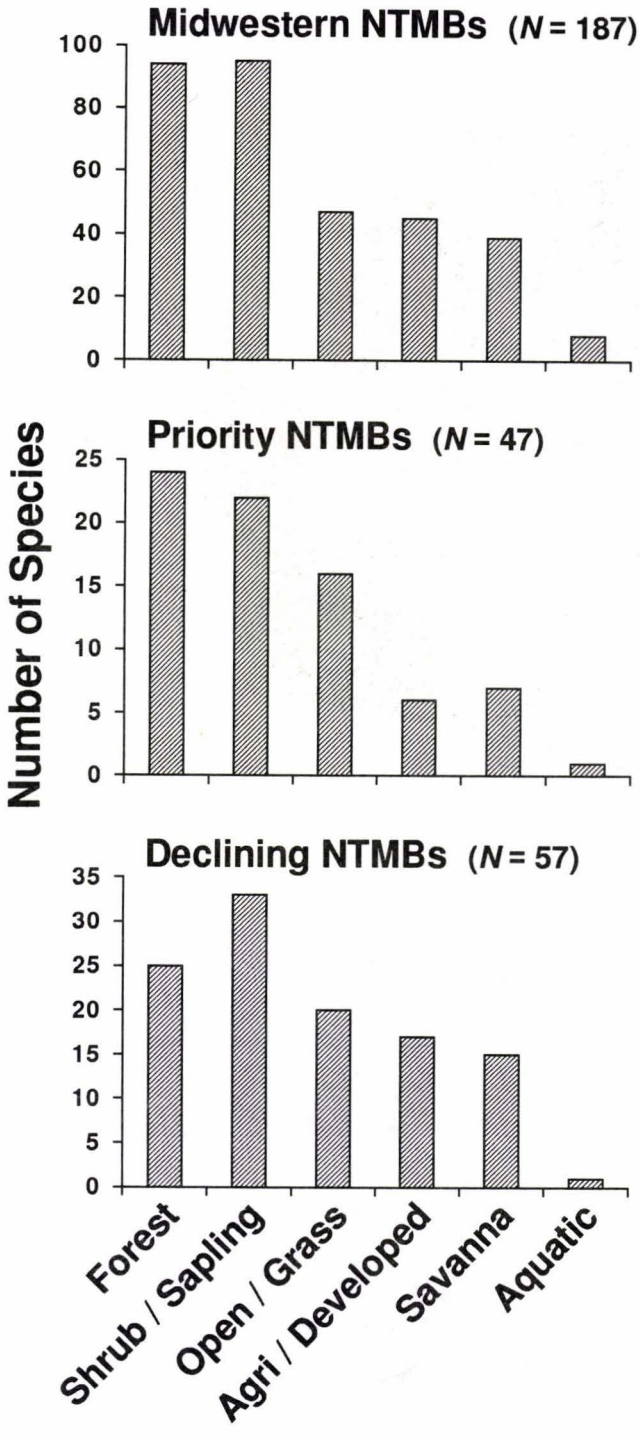


Figure 4.—*Number of Midwestern neotropical migratory birds, declining NTMBs, and priority NTMBs in five general land cover classes. Species may be associated with more than one land cover, so the bars may sum to more than the total number of species (N).*

15 percent with savanna, 13 percent with agricultural/developed, and 2 percent with aquatic habitats (fig. 4).

The number of NTMB species in land covers within Provinces ranged from 0 in aquatic land cover in the Ozark Broadleaf Forest-Meadow to 71 in forest in the Laurentian Forest. Number of priority species ranged from 0 in several land cover-Province combinations to 19 in Laurentian Forest Province. The distribution of species and priority species largely followed the expected distribution of land covers within provinces. For instance the highest numbers of priority species occurred within forests in the Laurentian Forest Province and within shrublands in the Eastern Broadleaf Forest Province and within grassland provinces in the Great Plains (table 1).

The finer breakdown of some land covers into habitat types revealed additional patterns. Within forested habitats, there were slightly more species, priority species, and declining species associated with deciduous than coniferous forests. More species were associated with upland forests than with lowland. More shrub species were associated with upland shrub-sapling habitats than with shrub wetland habitats (fig. 5). NTMBs, priority species, and declining species associated with agricultural habitats are largely dependent on pasture, hayfields, and fencerows as opposed to cropland (Koford and Best 1996). Indeed, the high relative proportion of priority and declining species in grasslands and agricultural habitats is a reflection of the lower proportion of grassland remaining relative to agricultural areas (fig. 3) converted from grassland, forest, or wetland.

Distribution of declining species among landcovers and habitats in many ways mirrored the distribution of species and priority species. Shrub and forest landcovers had the greatest number of declining species; savanna, grassland, and agriculture/developed landcovers had intermediate numbers, and aquatic landcovers had the fewest declining species (fig. 4). Caution should be used when interpreting these figures because some of these land covers may actually be ecological traps or sinks. For instance, many agricultural habitats may be ecological traps for grassland birds (Koford and Best 1996).

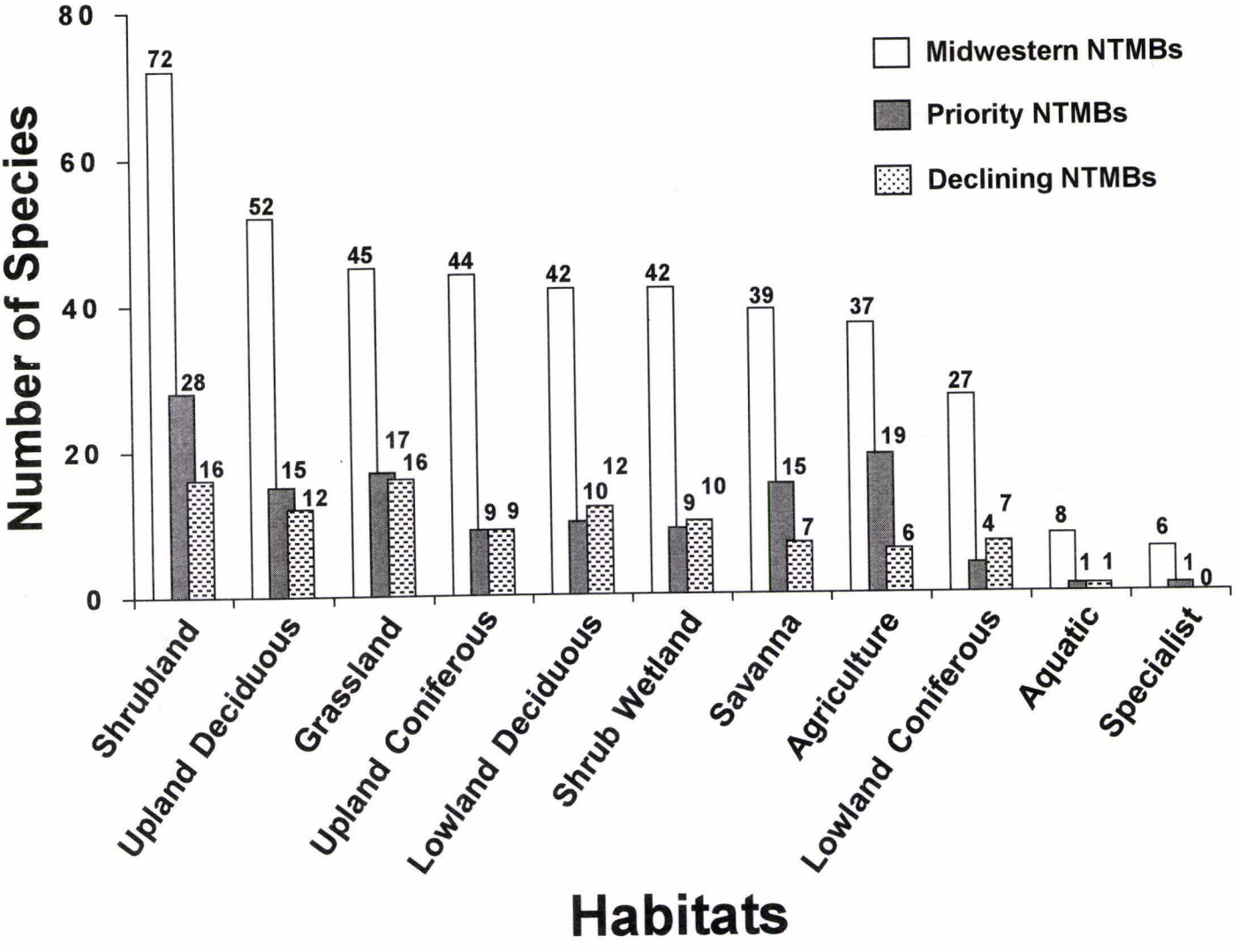


Figure 5.—*Number of neotropical migratory birds, high priority neotropical migratory birds, and declining neotropical migratory birds in Midwestern North America.*

Cover types, forest types, and their area and distribution are important determinants of animal distributions and populations. In addition, habitat age and age-distribution as affected by forest maturity and plant succession are also critical to landscape composition and structure, as well as to avian habitat associations. Midwestern NTMBs showed patterns among upland (dry) versus lowland (wet) ecosystems, conifer versus deciduous forests, and shrub/sapling versus mature forest, so we suggest the use of such gradients in future classification work (e.g., fig. 6). Thompson *et al.* (1996) show overlapping distributions of central hardwood birds across a disturbance gradient. These patterns illustrate that NTMB conservation is far broader than issues concerning forest birds and forest fragmentation. Gradients are easily related to maps of climate, landform, vegetation types, or tables of forest age distribution. Further, gradient approaches to classification are well-adapted to temporal change due to succession, climate, or land use modification. Thus, the general habitat area information can be modified to provide specific information on potential habitat area for species or species groups at several levels of resolution. We provide examples and applications of these types of habitat gradients for some grassland and forest NTMBs; full development of gradient classification is beyond the scope of this paper.

Geographic Links

Midwestern priority species have substantial geographic links to other regions of North America. Not surprisingly, many of the priority species in the Midwest are grassland species whose conservation must be coordinated with

Table 1.—*Numbers of midwestern neotropical migratory birds and priority species (in parentheses) that breed in land covers and ecological provinces. Species can be associated with more than one land cover, so rows and columns do not sum to species totals.*

Habitat	Ecological Province[1]							
	212	222	251	331	332	M222	M334	Total
Shrub/sapling	65 (14)	54 (16)	54 (11)	51 (8)	48 (8)	35 (8)	29 (3)	95 (22)
Forest	71 (19)	34 (10)	35 (6)	31 (3)	27 (4)	20 (5)	20 (2)	94 (24)
Agri./Developed	38 (5)	38 (5)	40 (6)	39 (6)	39 (6)	31 (4)	29 (4)	47 (6)
Grassland	26 (6)	25 (6)	39 (15)	35 (9)	33 (8)	20 (5)	21 (4)	45 (16)
Savanna	27 (6)	30 (6)	33 (6)	31 (3)	34 (5)	27 (5)	20 (2)	39 (7)
Aquatic	8 (1)	6 (1)	6 (1)	6 (1)	6 (1)	4 (1)	0 (0)	8 (1)
Total	129 (30)	124 (30)	136 (33)	126 (23)	125 (27)	93 (21)	81 (10)	187 (47)

[1]Based on Bailey *et al.* (1994) and McNab and Avers (1994); 212 = Laurentian Mixed Forest Province; 222 = Eastern Broadleaf Forest (continental); 251 = Prairie Parkland (Temperate); 331 = Great Plains Palouse Dry Steppe; 332 = Great Plains Steppe; M222 = Ozark Broadleaf Forest-Meadow; M334 = Black Hill Coniferous Forest.

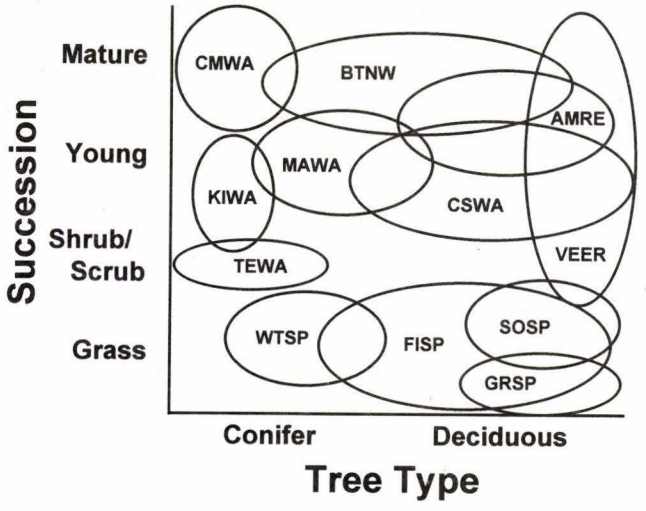


Figure 6.—*Hypothesized habitat relationships of some Midwestern neotropical migratory birds along gradients of forest seral stages and coniferous to deciduous tree life forms.*

Canada and the Western US (table 2). The Midwest is particularly important for the mountain plover, long-billed curlew, scissor-tailed flycatcher, Sprague's pipit, sedge wren, McCown's longspur, chestnut-collared longspur, and lark lunting. Species associated with deciduous forest are predominately linked to the Northeast (e.g., Canada warbler, black-throated blue warbler) and Southeast (e.g., Acadian flycatcher, wood thrush), which again is not suprising given the distribution of the eastern deciduous forest. Priority species associated with coniferous habitats were predominately linked to both the Northeast and Northwest, reflecting the boreal distribution of northern coniferous forests.

CONSERVATION PLANNING

We believe planning for optimal populations of migrant birds or other species requires spatial planning across a species range and integration with other resource values. A broad geographic perspective has several advantages: (1) Viability of individual populations is increased by allocating the largest possible area to each species and associated resource values such as other species or human uses. (2) Broad-based planning for integrated resource values minimizes resource conflicts by sorting resource objectives according to land capabilities and complementary ownership objectives. (3) Large-scale planning allows for consideration of trends in global change at multiple scales. Planning that is broad in scale and scope of issues can simplify legal compliance with laws such as NEPA of 1968, ESA of 1973, or NFMA of 1976 by simultaneous consideration of issues. Population processes can be assessed comprehensively and cost-effectively by a step-wise, successive approximation procedure that adds resolution in a strategic, systematic way (Freemark *et al.* 1993).

The geographical and ecological distribution information summarized here are examples of the types of data needed for Habitat Conservation Assessments (HCAs) to plan for both

Table 2.—*Overlap of priority species in the Midwest with other regions of North America. See text for methods used to identify priority species*

Species	NW	SW	NE	SE
Acadian flycatcher				X
Baird's sparrow	X			
Bay-breasted warbler	X		X	
Bell's vireo		X		
Black-billed cuckoo	X		X	
Black-throated blue warbler			X	
Blackburnian warbler			X	
Blue-winged warbler			X	X
Bobolink	X		X	
Burrowing Owl	X	X		
Canada warbler	X		X	
Cape May warbler	X		X	
Cassin's sparrow		X		
Cerulean warbler			X	
Chestnut-collared longspur	X			
Chestnut-sided warbler			X	
Clay-colored sparrow	X			
Connecticut warbler	X		X	
Dickcissel	X	X		X
Ferruginous hawk	X	X		
Golden-winged warbler			X	
Grasshopper sparrow	X	X	X	X
Gray-cheeked thrush	X		X	
Great crested flycatcher			X	X
Kirtland's warbler			X	
Lark bunting	X	X		
Loggerhead Shrike	X	X	X	X
Long-billed curlew	X	X		
Louisiana waterthrush			X	X
MacGillivray's warbler	X	X		
McCown's longspur	X	X		
Mississippi kite		X		X
Mountain plover	X	X		
Mourning warbler	X		X	
Nashville warbler	X	X	X	
Olive-sided flycatcher	X		X	
Painted bunting		X		
Philadelphia vireo	X		X	
Prairie warbler			X	X
Prothonotary warbler				X
Sedge wren	X		X	X
Sprague's pipit	X			
Swainson's warbler				X
Upland sandpiper	X	X	X	
Wood thrush			X	X
Worm-eating warbler			X	X
Yellow-billed cuckoo	X	X	X	X

viable species populations and ecologically effective populations (e.g., Connor 1988). Often, holistic management objectives require re-alignment of conventional approaches, including simultaneous planning for ecosystems as different as wetlands and grasslands. For example, at continental to landscape scales, conservation and management of wetlands, barrens and grasslands are often conveniently considered together. Not only do wetlands and grasslands or barrens frequently occur in the same landscapes, but they are often affected by agriculture at the same time and place. Prairie-wetland complexes may contain extreme moisture gradients, with overlapping bird species distributions along this and other gradients, which is a more general and dynamic way of classifying and assessing avian distribution. Single and multiple gradients such as this can help explain species distribution and abundance at scales from continental to local by accommodating variability within Ecological Units or vegetation zones.

At regional scales, the area and distribution of ecosystems (Howe *et al.* 1996) and their trends in vegetation, succession, land use, and landscape structure should be considered (Thompson *et al.* 1993). Geographic locations of productive sources should be emphasized as much as places where species are rare. It is critical to match landscape and local prescriptions to land capabilities and ownership objectives in a complementary manner. Much of the difference between Ecosystem Management and older concepts of multiple use involves planning in space and time rather than attempting to do all things in too small an area, or on lands with inappropriate capabilities. Thus, what we choose NOT to do in an ecosystem or ownership category may be as important as what we choose to do.

At subregional and human landscape scales, major considerations include distribution of forest types, forest age classes, and non-forest habitats within the context of ecosystem capabilities, disturbance frequency and pattern, and successional pathways (Thompson *et al.* 1993). At the level of administrative units within an ownership, the distribution of conditions in space and time becomes finer and stand-specific. Considerations include age classes of ecosystems since harvest or disturbance, as well as the mix and distribution of stand conditions such as vegetative

composition, vertical and horizontal structure of life forms, and special features such as dead and down material. At the stand level, silvicultural and rangeland prescriptions are chosen to achieve appropriate conditions for present and future landscape conditions within and across management units.

Conditions and cultural techniques can be chosen to emphasize, where appropriate, area-sensitive birds, cavity-nesters, canopy gleaners, understory gleaners, ground-foraging birds, or early succession species. In the past, coarse-filter approaches to landscape composition and structure have been used to try to provide for most species needs by creating a variety of ecosystems and conditions (e.g., Hunter *et al.* 1993, Hunter 1990, Crow *et al.* 1994). Matching species (and other resource objectives) to geographic and ecological distributions at broad scales is the first step to a finer filter for biodiversity and other more traditional human values for which we manage. Subsequent assessments at subregional and landscape scales for birds, other wildlife and plants, commodities, and other resource values will provide the necessary context for citizens, managers, and decision-makers to assess most cumulative and indirect effects in a more direct and reliable way.

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Appendix 1.—Neotropical migratory birds that occur in physiographic regions in Midwestern North America. Species are ranked by levels of decreasing management concern based on the Partners in Flight species prioritization scheme. Physiographic level scores were adjusted to reflect the Midwestern region of North America (see text for methods). Population trends are calculated from Breeding Bird Survey data for the period 1966-1994. Habitats are commonly used breeding habitats in the Midwest (see text for methods).

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4		
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.			
1	Kirtland's warbler	A	2	5.0	5.0	5.0	5.0	5.0	5.0	3.0	33.0	.		UCON	SHSA	
2	Baird's sparrow	A	4	4.0	4.5	4.0	4.5	4.0	5.0	3.8	29.8	-0.9		GRAS		
3	Mountain plover	A	2	4.0	5.0	3.0	5.0	4.0	5.0	3.0	29.0	-2.9	***	GRAS		
4	McCown's longspur	B	4	3.0	4.8	4.0	4.8	4.0	5.0	2.5	28.0	3.6		GRAS		
5	Chestnut-collared longspur	B	5	3.0	4.6	4.0	4.0	4.0	5.0	3.0	27.6	-0.3		GRAS		
6	Chestnut-sided warbler	A	10	3.0	4.0	4.0	3.0	5.0	5.0	2.7	26.7	-0.4		SHSA		
7	Golden-winged warbler	A	5	4.0	4.0	4.0	4.0	4.0	3.0	3.6	26.6	-3.6	***	SHSA	UDEC	
8	Sprague's pipit	B	4	3.5	3.0	3.0	4.0	3.5	5.0	3.8	25.8	-0.1		GRAS		
9	Cassin's sparrow	B	4	4.0	4.0	3.0	4.0	4.0	3.0	3.5	25.5	-2.4	***	SHSA	GRAS	
10	Connecticut warbler	A	4	4.0	3.0	3.0	4.0	4.0	5.0	2.5	25.5	3.0		LCON	SHWE	
11	Cerulean warbler	A	10	4.0	4.0	4.0	4.0	4.0	2.0	3.3	25.3	-4.3	***	LDEC	UDEC	
12	Black-billed cuckoo	A	21	3.0	3.0	3.8	3.0	4.0	5.0	3.3	25.1	-1.2	**	SHSA	SAVA	UDEC
13	Swainson's warbler	A	1	4.0	4.0	4.0	4.0	5.0	1.0	3.0	25.0	1.6		SHWE	LDEC	
14	Long-billed curlew	A	7	3.0	3.6	3.0	4.0	4.0	4.0	3.3	24.9	-1.3		GRAS		
15	Blackburnian warbler	A	6	3.0	4.0	4.0	3.0	4.0	4.0	2.8	24.8	0.6		UCON	LCON	
16	Sedge wren	B	16	4.0	4.0	3.0	3.0	3.8	4.0	3.1	24.8	1.7	*	SHWE	AQUA	GRAS
17	Philadelphia vireo	A	4	4.0	3.0	4.0	3.0	5.0	3.0	2.5	24.5	4.2		SHSA	SHWE	SAVA
18	Clay-colored sparrow	A	14	3.0	3.7	2.9	3.1	3.1	5.0	3.4	24.2	-1.3		SHSA	SAVA	
19	Bay-breasted warbler	A	3	2.0	2.0	4.0	3.0	5.0	5.0	3.0	24.0	-0.2		UCON	LCON	
20	Black-throated blue warbler	A	5	3.0	4.0	4.0	3.0	4.0	3.0	3.0	24.0	0.6		UDEC	LDEC	
21	Lark bunting	A	9	2.0	4.0	3.0	3.4	3.0	5.0	3.6	24.0	-0.7		GRAS		
22	Mississippi kite	A	6	3.0	3.0	3.0	3.0	4.0	5.0	3.0	24.0	0.1		UDEC	LDEC	SAVA
23	Bobolink	A	18	2.0	4.0	3.0	3.0	3.2	5.0	3.8	23.9	-1.8	***	GRAS	AGRI	
24	Ferruginous hawk	B	7	4.0	4.0	3.0	3.0	3.0	4.0	2.7	23.7	4.7	**	GRAS		
25	Bell's vireo	A	13	3.0	4.2	3.2	3.0	4.0	3.0	3.4	23.7	-3.0	**	SHWE	SHSA	
26	Dickcissel	A	19	2.0	4.0	2.8	3.0	4.0	4.0	3.8	23.6	-1.6	***	GRAS	AGRI	
27	Yellow-billed cuckoo	A	20	3.0	3.4	3.8	2.0	3.0	5.0	3.3	23.5	-1.4	***	UDEC	LDEC	

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4			
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.				
28	Mourning warbler	A	9	3.0	3.0	3.0	3.0	4.0	5.0	2.4	23.4	1.3	*	SHSA	SHWE	UDEC	UCON
29	Gray-cheeked thrush	A	3	3.0	2.3	3.0	3.0	4.0	5.0	3.0	23.3	.		UCON	LCON		
30	Wood thrush	A	16	2.0	4.0	4.0	3.0	4.0	3.0	3.3	23.3	-1.7	***	LDEC	UDEC		
31	Painted bunting	A	4	3.0	4.0	3.0	4.0	3.0	3.0	3.3	23.3	-3.3	***	SHSA			
32	Grasshopper sparrow	A	20	2.2	4.0	2.8	2.0	3.0	5.0	4.2	23.2	-3.7	***	GRAS	AGRI		
33	Canada warbler	A	6	2.0	4.0	4.0	3.0	4.0	3.0	3.2	23.2	-3.1	**	UCON	UDEC	SHWE	
34	Louisiana waterthrush	A	9	4.0	4.0	4.0	3.0	3.0	2.0	3.1	23.1	0.3		LDEC			
35	Great crested flycatcher	A	19	2.0	3.0	4.0	3.0	4.0	4.0	3.1	23.1	-0.2		UDEC	LDEC		
36	MacGillivray's warbler	A	3	3.0	3.0	3.0	4.0	4.0	3.0	3.0	23.0	-1.0		SHWE	LCON	LDEC	SHSA
37	Worm-eating warbler	A	5	3.0	4.0	4.0	3.0	4.0	2.0	3.0	23.0	1.0		UDEC	LDEC	SHSA	
38	Loggerhead shrike	B	21	3.0	4.2	4.0	2.0	3.0	3.0	3.6	22.8	-3.5	***	SAVA	SHSA	GRAS	AGRI
39	Blue-winged warbler	A	11	3.0	3.0	4.0	4.0	4.0	2.0	2.7	22.7	0.4		SHSA	SHWE	SAVA	
40	Prothonotary warbler	A	7	3.0	4.0	4.0	3.0	4.0	2.0	2.7	22.7	-1.6	**	LDEC	SHWE		
41	Cape May warbler	A	5	3.0	2.0	4.0	3.0	4.0	4.0	2.6	22.6	-7.9		UCON	LCON		
42	Nashville warbler	A	7	3.0	3.0	3.0	3.0	4.0	4.0	2.6	22.6	0.1		SHSA	UCON	LCON	
43	Olive-sided flycatcher	A	11	3.0	3.2	4.0	2.2	3.2	4.0	2.9	22.5	-4.1	***	UCON	SHSA		
44	Prairie warbler	A	11	2.0	4.0	3.0	3.0	4.0	3.0	3.5	22.5	-2.7	***	SHSA	SAVA		
45	Upland sandpiper	A	18	3.0	3.0	3.0	3.0	3.0	5.0	2.4	22.4	2.0	***	GRAS	AGRI		
46	Acadian flycatcher	A	9	2.0	3.0	4.0	3.0	4.0	3.0	3.2	22.2	0.5		LDEC	UDEC		
47	Burrowing owl	A	9	4.0	4.3	3.0	1.7	1.7	4.0	3.6	22.2	-0.5		GRAS	AGRI		
48	Ovenbird	A	19	2.0	3.2	3.2	3.0	3.0	5.0	2.7	22.2	1.2	***	UDEC	LDEC		
49	Veery	A	13	2.0	3.0	3.0	3.0	3.0	5.0	3.1	22.1	-1.1	***	LDEC	SHSA		
50	Hooded warbler	A	8	3.0	3.0	4.0	3.0	4.0	2.0	3.0	22.0	0.4		LDEC	UDEC	SHSA	
51	Lewis' woodpecker	B	3	2.0	4.0	3.0	4.0	3.0	3.0	3.0	22.0	-3.4	**	SAVA			
52	Northern parula	A	10	2.0	4.0	3.0	3.0	3.0	4.0	3.0	22.0	0.1		LDEC	LCON	UCON	
53	Gray catbird	A	21	2.0	2.8	3.6	2.0	3.0	5.0	3.5	21.9	-0.2		SHSA	SHWE		
54	Tennessee warbler	A	7	3.0	2.1	3.0	3.0	4.0	4.0	2.7	21.9	6.3		UCON	LCON	UDEC	LDEC
55	Orchard oriole	A	18	3.0	3.2	2.8	3.0	3.0	4.0	2.8	21.8	-1.9	***	SAVA	AGRI	DEVE	
56	Blackpoll warbler	A	4	2.0	2.3	3.0	3.0	4.0	5.0	2.5	21.8	-0.2		UCON	LCON	SHSA	
57	Scissor-tailed flycatcher	A	6	2.0	2.0	2.0	4.0	4.0	4.0	3.7	21.7	0.1		GRAS	AGRI	SHSA	SAVA

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4				
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.					
58	Rose-breasted grosbeak	A	19	2.0	3.0	2.8	3.0	3.0	5.0	2.8	21.6	0.0		UDEC	SAVA			
59	Scarlet tanager	A	14	2.0	3.0	4.0	3.0	4.0	3.0	2.6	21.6	0.2		UDEC	LDEC			
60	Yellow-throated vireo	A	14	3.0	3.0	4.0	3.0	3.0	3.0	2.5	21.5	1.1	***	LDEC	UDEC			
61	Eastern wood-pewee	A	17	2.0	2.0	3.0	3.0	4.0	4.0	3.4	21.4	-1.7	***	UDEC				
62	Kentucky warbler	A	7	2.0	3.0	3.0	3.0	4.0	3.0	3.3	21.3	-1.0	*	LDEC	UDEC	SHSA		
63	Merlin	A	10	4.0	2.4	3.0	2.0	2.0	5.0	2.8	21.2	.		UCON	LCON	GRAS	SHSA	SHWE
64	Yellow-headed blackbird	A	15	2.2	3.2	3.0	3.0	3.0	4.0	2.8	21.2	0.2		AQUA	SHWE			
65	Allen's hummingbird	A	1	2.0	3.0	3.0	5.0	4.0	1.0	3.0	21.0	-1.5		SHSA	SHWE			
66	Black-throated gray warbler	A	1	3.0	4.0	3.0	3.0	4.0	1.0	3.0	21.0	1.3		UCON	UDEC			
67	Cordilleran flycatcher	A	2	2.0	3.0	3.0	4.0	4.0	2.0	3.0	21.0	.		UCON	LCON			
68	Dusky flycatcher	A	2	2.0	3.0	3.0	4.0	4.0	2.0	3.0	21.0	3.0	*	UDEC	UCON	SHSA		
69	Hammond's flycatcher	A	1	2.0	3.0	4.0	4.0	4.0	1.0	3.0	21.0	0.7		UCON				
70	Northern goshawk	B	8	3.0	4.0	3.0	2.0	2.0	4.0	3.0	21.0	.		UCON	UDEC			
71	Prairie falcon	B	6	3.0	4.0	3.0	3.0	2.0	3.0	3.0	21.0	0.3		GRAS				
72	Red-naped sapsucker	B	20	2.0	3.0	4.0	4.0	3.0	2.0	3.0	21.0	.		UCON				
73	Sage sparrow	B	2	3.0	3.0	3.0	4.0	4.0	1.0	3.0	21.0	-0.5		SAVA	GRAS			
74	Willow flycatcher	A	12	3.0	3.7	3.0	2.3	3.0	3.0	2.8	20.8	.		SHSA	SHWE			
75	Least flycatcher	A	16	3.0	3.3	2.5	2.0	3.0	4.0	3.1	20.8	-1.6	***	UDEC	LDEC			
76	Marsh wren	B	17	2.0	4.0	4.0	2.2	1.8	4.0	2.8	20.8	2.6		SHWE	AQUA			
77	Brewer's sparrow	A	5	2.0	3.0	3.6	3.0	3.0	3.0	3.0	20.6	-3.7	***	SHSA	SAVA			
78	Black-throated green warbler	A	6	3.0	3.0	3.0	3.0	3.0	3.0	2.5	20.5	-0.2		UCON	UDEC			
79	Magnolia warbler	A	6	3.0	2.0	3.0	3.0	3.0	4.0	2.5	20.5	2.3	**	UCON	LCON	SHSA		
80	Northern harrier	B	19	4.0	4.0	4.0	1.0	1.0	3.0	3.5	20.5	-1.2	*	GRAS	SAVA	SHWE	AGRI	
81	Swainson's hawk	A	10	3.0	2.7	3.0	2.0	3.0	4.0	2.7	20.4	1.8	**	GRAS	AGRI			
82	Cassin's kingbird	A	3	3.0	3.0	2.3	3.0	4.0	2.0	3.0	20.3	-1.9		GRAS	SAVA	SHSA		
83	Peregrine falcon	A	6	4.0	4.3	4.0	1.7	1.3	2.0	3.0	20.3	.		SPEC				
84	Whip-poor-will	A	12	2.0	3.0	3.0	3.0	3.0	3.0	3.3	20.3	-1.2	*	UDEC				
85	Yellow-bellied flycatcher	A	5	2.0	2.0	4.0	3.0	4.0	3.0	2.2	20.2	0.1		UCON	SHWE			
86	Short-eared owl	B	20	4.0	4.0	4.0	1.0	1.0	3.0	3.2	20.2	1.9		GRAS	SAVA			
87	Western kingbird	A	15	2.0	1.7	2.7	3.0	4.0	4.0	2.7	20.1	0.9	***	SAVA	AGRI			

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4			
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.				
88	Summer tanager	A	8	2.0	3.0	3.0	3.0	3.0	3.0	3.1	20.1	-0.2		UDEC	SAVA		
89	Alder flycatcher	A	8	3.0	3.0	3.0	2.0	4.0	2.0	3.0	20.0	.		SHWE			
90	Gray kingbird	D	1	2.0	3.0	3.0	4.0	4.0	1.0	3.0	20.0	.					
91	Lark sparrow	A	17	3.0	2.8	2.8	2.0	3.0	3.0	3.4	19.9	-3.4	***	SHSA	GRAS	AGRI	
92	Red-shouldered hawk	B	11	3.0	3.0	3.0	3.0	3.0	2.0	2.9	19.9	1.6	**	LDEC	SHWE		
93	Eastern kingbird	A	22	2.0	2.0	3.0	2.0	3.0	5.0	2.9	19.9	-0.5	**	SHSA	SAVA	GRAS	AGRI
94	Purple finch	B	7	3.0	3.0	2.0	3.0	3.0	3.0	2.7	19.7	-1.0	**	UCON	SHSA	UDEC	
95	Western meadowlark	B	19	1.0	3.0	3.0	2.0	1.8	5.0	3.9	19.7	-0.6	*	GRAS	AGRI		
96	Palm warbler	A	3	3.0	2.0	2.0	3.0	3.0	4.0	2.7	19.7	.		LCON	SHWE		
97	Golden-crowned kinglet	B	8	3.0	3.0	2.0	3.0	2.0	4.0	2.5	19.5	-3.3	***	UCON	LCON		
98	Yellow-breasted chat	A	18	2.0	3.0	3.0	2.0	3.0	3.0	3.5	19.5	-0.5	*	SHSA			
99	Yellow-throated warbler	A	6	3.0	3.0	3.0	3.0	3.0	2.0	2.5	19.5	0.8		UDEC	LDEC		
100	Northern oriole	A	20	2.0	3.0	2.8	3.0	3.0	3.0	2.7	19.5	-0.4	*	SAVA	AGRI	DEVE	
101	Black-and-white warbler	A	16	2.0	4.0	2.8	2.0	2.0	4.0	2.6	19.4	-1.2	***	UDEC	LDEC		
102	Vesper sparrow	B	21	3.0	2.0	2.0	2.0	2.0	5.0	3.4	19.4	-0.9	**	SHSA	SAVA	GRAS	AGRI
103	White-throated sparrow	B	8	1.9	2.0	1.0	3.0	3.0	5.0	3.5	19.4	-1.1	**	SHSA	SHWE	LCON	
104	Mountain bluebird	B	6	2.0	3.0	3.0	3.0	2.3	3.0	3.0	19.3	.		UCON	UDEC	SHSA	
105	Swainson's thrush	A	10	3.0	3.0	3.4	2.0	2.0	3.0	2.9	19.3	-0.8	*	UCON	LCON	SHSA	
106	Wilson's warbler	A	7	3.0	2.3	2.0	2.3	3.0	4.0	2.7	19.3	-0.2		SHWE	SHSA		
107	Lazuli bunting	A	4	2.0	3.0	2.0	3.0	4.0	2.0	3.3	19.3	-0.9		SHSA	SHWE		
108	Solitary vireo	A	8	3.0	3.3	3.0	2.8	3.0	2.0	2.3	19.3	3.3	***	UCON	UDEC		
109	White-eyed vireo	A	8	2.0	3.0	3.0	3.0	3.0	2.0	3.3	19.3	0.0		SHSA	SAVA	SHWE	
110	Indigo bunting	A	21	1.0	2.0	2.0	3.0	3.0	5.0	3.2	19.2	-0.7	***	SHSA	SAVA	SHWE	
111	American redstart	A	19	2.0	2.7	3.0	2.0	2.3	4.0	3.1	19.1	-0.7		UDEC	LDEC		
112	Common poorwill	B	4	3.0	2.0	3.0	3.0	3.0	2.0	3.0	19.0	3.5		SAVA	SHSA	GRAS	
113	Green-tailed towhee	A	2	3.0	2.0	2.0	4.0	3.0	2.0	3.0	19.0	-0.3		SHSA			
114	Purple martin	A	20	2.0	3.0	3.0	2.0	3.0	3.0	3.0	19.0	0.0		AQUA	DEVE	AGRI	
115	Townsend's solitaire	B	2	3.0	3.0	3.0	3.0	3.0	1.0	3.0	19.0	1.5		UCON	SHSA		
116	Cooper's hawk	B	21	3.0	4.0	3.0	1.0	2.0	3.0	2.9	18.9	6.8	***	UCON	LCON	UDEC	LDEC
117	Sharp-shinned hawk	B	20	3.0	4.0	3.0	1.0	1.0	4.0	2.9	18.9	-0.2		UCON	UDEC		

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4				
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.					
118	Broad-winged hawk	A	14	3.0	3.0	3.0	2.0	2.0	3.0	2.9	18.9	0.8		UDEC	LDEC			
119	Black-headed grosbeak	A	5	2.0	3.0	2.2	3.0	4.0	2.0	2.6	18.8	0.0		UDEC	LDEC	SHSA		
120	Hermit thrush	B	9	3.0	3.0	3.0	2.0	2.0	3.0	2.8	18.8	1.7	***	UCON	LCON	SHSA	SHWE	
121	Chuck-will's-widow	A	8	2.0	3.0	3.0	3.0	3.0	2.0	2.8	18.8	-1.4	***	UDEC	UCON			
122	Osprey	B	11	3.0	4.0	3.0	1.0	1.0	4.0	2.7	18.7	6.8	***	AQUA	LDEC	LCON		
123	Chimney swift	A	21	1.0	2.6	2.8	2.0	3.0	4.0	3.2	18.7	-0.9	***	SPEC				
124	Sage thrasher	B	2	2.0	2.0	2.0	4.0	3.0	3.0	2.5	18.5	0.6		SHSA				
125	Bank swallow	A	21	3.0	2.2	2.0	1.0	3.0	4.0	3.2	18.4	-1.4		AQUA				
126	Western tanager	A	3	2.0	3.0	2.3	3.0	3.0	2.0	3.0	18.3	0.0		UCON	LCON	UDEC		
127	Rock wren	B	5	3.0	2.0	2.0	2.8	2.2	3.0	3.2	18.2	-1.0		SPEC				
128	Blue grosbeak	A	12	3.0	2.2	2.0	2.0	3.0	3.0	2.9	18.1	1.5	***	SHSA	SAVA	SHWE	AGRI	
129	American pipit	B	3	2.0	2.0	3.0	3.0	2.0	3.0	3.0	18.0	.		GRAS				
130	Black-throated sparrow	B	1	3.0	2.0	3.0	3.0	3.0	1.0	3.0	18.0	-3.9	**	GRAS				
131	Northern waterthrush	A	9	2.0	2.1	3.1	2.0	2.0	4.0	2.8	18.0	0.0		LDEC	SHWE			
132	Orange-crowned warbler	A	5	3.0	2.0	2.0	2.4	3.0	3.0	2.6	18.0	-1.0		SHSA	SHWE			
133	Violet-green swallow	A	4	2.0	3.0	2.0	3.0	3.0	2.0	3.0	18.0	-0.9	*	UCON	UDEC			
134	White-throated swift	A	3	3.0	2.0	2.0	3.0	3.0	2.0	3.0	18.0	-2.3		SPEC				
135	Yellow-bellied sapsucker	B	12	2.0	2.0	3.0	3.0	2.0	3.0	3.0	18.0	.		UDEC				
136	Eastern phoebe	B	21	2.0	3.0	3.0	2.0	2.0	3.0	2.9	17.9	0.9	***	SPEC				
137	Eastern bluebird	B	22	2.0	3.0	3.0	2.2	2.0	3.0	2.6	17.8	2.4	***	SAVA	AGRI	SHSA	GRAS	
138	Say's phoebe	B	8	3.0	2.0	3.0	2.0	3.0	2.0	2.8	17.8	0.8		SHSA	SAVA	GRAS	AGRI	
139	Western wood-pewee	A	9	2.0	2.0	3.0	2.0	4.0	2.0	2.7	17.7	-2.0	***	UDEC	UCON	SHSA		
140	Savannah sparrow	B	18	2.2	3.0	3.0	1.0	2.0	3.0	3.3	17.6	-1.2	***	GRAS	AGRI			
141	Swamp sparrow	B	12	2.8	1.8	1.8	2.8	2.8	3.0	2.6	17.5	1.9	***	SHWE				
142	Tree swallow	B	20	2.0	3.0	3.0	1.0	2.0	4.0	2.3	17.3	1.3	***	SAVA	SHWE	AQUA		
143	Warbling vireo	A	21	2.0	2.4	2.8	1.2	3.0	3.0	2.7	17.1	0.3		LDEC	LCON	SAVA	SHSA	
144	Black-chinned hummingbird	A	1	2.0	2.0	2.0	3.0	4.0	1.0	3.0	17.0	1.2		UDEC	LDEC			
145	Cassin's finch	B	1	2.0	3.0	2.0	3.0	2.0	2.0	3.0	17.0	0.4		UCON				
146	Golden eagle	B	5	3.0	2.0	2.0	2.0	2.0	3.0	3.0	17.0	2.1		GRAS				
147	Ruby-throated hummingbird	A	18	2.0	2.0	2.0	2.0	3.0	3.0	2.9	16.9	1.2	**	UDEC	LDEC	UCON	LCON	SHSA

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4			
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.				
148	Brewer's blackbird	B	16	2.0	2.0	2.0	3.0	2.0	3.0	2.9	16.9	-1.6	***	SHWE	SHSA	AGRI	LDEC
149	Blue-gray gnatcatcher	A	12	2.0	3.0	2.0	2.0	3.0	2.0	2.8	16.8	1.2	**	UDEC	LDEC	UCON	SHSA
150	Northern rough-winged swallow	A	21	3.0	2.2	2.0	1.0	3.0	3.0	2.4	16.6	-0.5		SAVA	SHWE	GRAS	
151	Common nighthawk	A	22	2.0	3.0	2.0	1.0	1.8	4.0	2.8	16.6	-0.9		SAVA	GRAS	AGRI	DEVE
152	Lincoln's sparrow	A	7	2.0	2.0	2.0	2.3	2.0	4.0	2.3	16.6	1.6		SHWE	SHSA		
153	Fox sparrow	B	4	3.0	2.0	2.0	2.0	2.0	3.0	2.5	16.5	0.8		LDEC	LCON	SHWE	
154	Rufous-sided towhee	B	20	1.0	3.0	2.0	2.0	2.0	3.0	3.5	16.5	-1.9	***	SHSA	SAVA		
155	Cedar waxwing	B	21	2.0	2.0	2.0	2.0	2.0	4.0	2.3	16.3	1.6	***	SHSA	SHWE	DEVE	UCON
156	Eastern meadowlark	B	16	1.0	3.0	3.0	1.0	1.0	3.0	4.3	16.3	-2.6	***	GRAS	AGRI		
157	Ruby-crowned kinglet	B	8	2.0	2.0	3.0	2.0	2.0	2.0	3.3	16.3	3.7		UCON	LCON	SHSA	
158	Ash-throated flycatcher	A	1	2.0	2.0	2.0	3.0	3.0	1.0	3.0	16.0	1.9	**	LDEC			
159	Lesser goldfinch	B	1	2.0	2.0	2.0	3.0	3.0	1.0	3.0	16.0	-2.1	**	SAVA	SHSA	AGRI	DEVE
160	Cliff swallow	A	22	2.0	2.2	2.0	1.0	3.0	3.0	2.7	16.0	0.8	*	SPEC			
161	Red-eyed vireo	A	22	1.0	2.2	2.0	2.0	3.0	3.0	2.7	16.0	1.0	***	UDEC	LDEC		
162	Long-eared owl	B	17	3.0	2.0	3.0	1.0	1.0	2.0	3.0	15.0	.		UCON	UDEC	AGRI	
163	Vermilion flycatcher	A	1	3.0	3.0	3.0	1.0	1.0	1.0	3.0	15.0	-2.9	*	LDEC	SHSA	SHWE	
164	White-crowned sparrow	B	5	1.6	2.0	1.0	2.0	2.0	3.0	3.0	14.6	-2.6	***	SHSA	SAVA		
165	Common yellowthroat	A	21	1.0	2.4	2.0	1.0	2.0	3.0	3.0	14.4	-0.2		SHSA	SHWE		
166	Belted kingfisher	B	22	2.0	2.0	2.0	1.0	1.0	3.0	3.2	14.2	-1.3	***	AQUA			
167	Chipping sparrow	A	22	1.0	2.2	2.0	1.0	2.0	3.0	2.8	14.0	-0.1		SAVA	SHSA		
168	Yellow warbler	A	22	1.0	3.2	2.0	1.0	1.0	3.0	2.7	14.0	0.6	**	SHWE			
169	American goldfinch	B	20	1.0	2.0	1.0	1.9	1.9	3.0	3.1	13.8	-1.1	***	SHSA	AGRI		
170	Northern mockingbird	B	17	1.9	1.0	1.0	2.0	2.0	2.0	3.5	13.4	-1.0	***	SHSA	DEVE	AGRI	
171	Brown creeper	B	11	2.0	2.0	2.0	1.0	1.0	2.0	3.1	13.1	-1.0		UDEC	LDEC		
172	Song sparrow	B	21	1.0	3.0	1.0	1.0	1.0	3.0	3.0	13.0	-0.2		SHSA	SHWE		
173	Brown-headed cowbird	B	21	1.0	1.0	1.0	1.0	2.0	3.0	3.5	12.5	-0.7	***	AGRI	GRAS	SAVA	
174	Red-winged blackbird	B	22	1.0	2.0	1.0	1.0	1.0	3.0	3.4	12.4	-1.2	***	SHWE	GRAS		
175	American kestrel	B	22	1.0	1.8	2.0	1.0	1.0	3.0	2.5	12.3	0.2		AGRI	SAVA		
176	Yellow-rumped warbler	B	11	1.0	2.0	1.0	1.0	1.3	3.0	3.0	12.3	3.7	***	UCON	LCON		
177	Northern flicker	B	22	1.0	1.0	1.0	1.0	1.0	3.0	4.1	12.1	.		SAVA	DEVE		

Appendix 1. Continued.

Rank	Species	Type1	PIF Scores2									Popn. trend3		Habitats4			
			No.	AB	TB	TW	BD	WD	IA	TR	Prior.	Trend	Signif.				
178	Turkey vulture	B	18	1.0	2.0	2.0	1.0	1.0	3.0	2.1	12.1	0.9	*	UDEC	LDEC	AGRI	
179	Dark-eyed junco	B	9	1.0	2.0	1.0	1.0	1.0	3.0	2.9	11.9	-0.6		UCON	SHSA		
180	Pine siskin	B	12	1.0	1.0	1.0	2.0	1.0	3.0	2.8	11.8	-2.5	*	UCON	LCON	SHSA	
181	Barn swallow	A	22	1.0	1.0	2.0	1.0	1.0	3.0	2.6	11.6	0.3	*	AGRI	GRAS	DEVE	
182	Horned lark	B	22	1.0	1.0	1.0	1.0	1.0	3.0	3.3	11.3	-0.8	**	GRAS	AGRI		
183	Killdeer	B	22	1.0	1.0	1.0	1.0	1.0	3.0	2.6	10.6	0.1		GRAS	AGRI	DEVE	
184	Mourning dove	B	21	1.0	1.0	1.0	1.0	1.0	3.0	2.5	10.5	-0.2		SAVA	GRAS	AGRI	SHSA
185	Red-tailed hawk	B	22	1.0	1.0	1.2	1.0	1.0	3.0	2.3	10.5	3.2	***	UDEC	AGRI	GRAS	SHSA
186	House wren	A	21	1.0	1.0	1.0	1.0	1.0	3.0	2.1	10.1	1.4	***	UDEC	LDEC	DEVE	
187	American robin	B	22	1.0	1.0	1.0	1.0	1.0	3.0	1.7	9.7	0.9	***	DEVE	AGRI	UDEC	

[1]A = species winters primarily south of the U.S.; B = species winters primarily in the U.S. but a portion of population winters south of the U.S.

[2]Partners in Flight species priority scores. No. = number of physiographic regions species was included in; AB = global abundance; TB = threats on breeding grounds; TW = threats on wintering grounds; BD = breeding distribution; WD = wintering distribution; IA = importance of area; TR = 26-year population trend; Prior. = species priority score. Score of 1 for all criteria indicates least concern, 5 indicates greatest concern; see text and references for details.

[3]Trend is percent annual change for period 1966-1994. Significance values are: ***, $p<0.01$; **$p<0.05$; *, $p<0.1$. Estimates based on Breeding Bird Survey (information provided by Bruce G. Peterjohn, National Biological Service, Patuxent Environmental Science Center, Laurel, Maryland).

[4]Breeding habits. AQUA = aquatic; AGRI = agriculture, DEVE = developed; GRAS = grassland; SHSA = shrub-sapling; SHWE = shrub-wetland; SAVA = savanna; UDEC = upland deciduous forest; UCON = upland conifer forest; LDEC = lowland deciduous forest; LCON = lowland conifer forest; see text for habitat descriptions.

METAPOPULATIONS, SOURCES AND SINKS, AND THE CONSERVATION OF NEOTROPICAL MIGRATORY BIRDS IN THE MIDWEST

Therese M. Donovan[1], Douglas A. Clark[2], Robert W. Howe[3], and Brent J. Danielson[4]

ABSTRACT.—Metapopulation models and source-sink models depict species whose distributions are spatially subdivided at some scale. Although the distribution is spatially subdivided, the subdivided populations (also called subpopulations) are linked to each other when individuals disperse from one location to another. Both models examine the consequences of dispersal among subpopulations and its impact on the survival of the entire population. These models are relevant for migratory bird management because distribution gaps and dispersal are typical of many migratory bird populations; management at the local scale may have limited or perhaps no effect on global populations of priority species.

A metapopulation is a set of connected subpopulations where each subpopulation experiences extinction but is recolonized when individuals disperse from one subpopulation to another. Metapopulation models explain and predict the distribution of occupied and unoccupied habitat patches and factors that affect dispersal between patches and the persistence of the greater metapopulation. In a metapopulation context, critical information consists simply of presence or absence of a species for each patch across time. Factors that influence extinction and recolonization rates are studied to understand their impacts on the spatial distribution and persistence of the populations.

Source-sink models, by contrast, consider both the distribution and size of connected populations, where subpopulations vary in their birth and death rates. Sink subpopulations inhabit relatively poor quality habitat and are not viable without an influx of immigrants. Source subpopulations inhabit relatively high quality habitat in terms of reproduction and survival and are viable without an influx of immigrants; sources can export individuals to sinks. The numbers of individuals located within each subpopulation, the birth and death rates of each subpopulation, and rates of dispersal between subpopulations are critical in determining the size (number) of the entire population.

Although today's theoretical models describe many types of dynamics, their implications for the management of migratory birds are the same: (1) isolated breeding subpopulations are linked by dispersal, which may be critical for the maintenance of regional populations, (2) the spatial scale at which subpopulations interact by dispersal, while typically unknown, defines an appropriate demographic unit for management, (3) in the absence of dispersal data, regional-scale management is preferred over local-scale management, and (4) identifying and maintaining source subpopulations is vital for the persistence of spatially subdivided populations.

[1]*USDA North Central Forest Experiment Station, 1-26 Agriculture Bldg., University of Missouri, Columbia, MO 65211*

[2]*Learning Center Mathematics Division, University of Missouri, Columbia, MO 65211*

[3]*Department of Biology, University of Wisconsin - Green Bay, Green Bay, WI 54311-7001*

[4]*Department of Animal Ecology, Iowa State University, Ames, Iowa, 50011-3221*

INTRODUCTION

Many bird species have distributions that are geographically discontinuous; that is, they consist of multiple populations (Andrewartha and Birch 1954, den Boer 1981). From a management perspective, this discontinuity can influence whether or not a local population and the global population can persist over time. Metapopulation and source-sink models depict local populations (or subpopulations) that vary in longevity and emphasize the importance of dispersal among local populations (Harrison 1994). These models are relevant for migratory bird management because migratory bird populations are often spatially discontinuous but linked by dispersing individuals. Therefore, setting management goals directed solely at the local scale may have limited success because populations may be regulated by events occurring at scales beyond the local scale.

Our objectives are to: (1) briefly review the importance of metapopulation and source-sink concepts, (2) provide empirical evidence that supports or refutes these models, (3) discuss the demographic implications of each model, and (4) provide some general management recommendations in light of current knowledge of these systems. Our intent is to help managers understand the concepts and implications of metapopulation and source-sink theory for the management of migratory bird species. Based on empirical knowledge, managers can apply these concepts and assess appropriate actions on a species-by-species basis.

METAPOPULATIONS: CONCEPTS AND EMPIRICAL EVIDENCE

Metapopulation Concepts

The metapopulation model (Levins 1968, 1969, 1970) was originally devised to describe organisms in spatially discrete habitat patches where local subpopulations periodically go extinct and are recolonized; the persistence of the greater population depends on how the patches interact in terms of dispersal and local extinction (reviewed by Hanski and Gilpin 1991). Levins (1968, 1969, 1970) developed the "classic" metapopulation model to discern how a patchily distributed insect pest population could be controlled when it consisted of multiple populations. His concern was that local populations could be sprayed to extinction with seemingly little effect on the larger population, and thus he pursued the demographic consequences of localized extinction and recolon-ization in a "population of connected subpopulations."

In Levins' metapopulation model, a "metapopulation" exists in a network of habitat patches, some occupied by subpopulations of individuals and some unoccupied (fig. 1). Each subpopulation has a finite lifetime and each has the same probability of extinction. Although any local subpopulation can go extinct, the patch also can be recolonized by individuals dispersing from other subpopulations. Levins' model implicitly assumes that patches are of equal population size and contribute colonists equally. Each unoccupied patch has the same probability of being colonized. At equilibrium, the proportion of patches that are occupied remains constant, although the pattern of occupancy continually shifts as some subpopulations suffer extinction followed by recolonization (fig. 1).

Levins' model was a differential equation that determined the rate of change in the percentage of inhabited patches. There are two key parameters: the local extinction rate and the migration rate of individuals to other patches. The original equation is

$$dp/dt = mp(1-p) - ep,$$

where p is the proportion of inhabited patches, m is the migration rate (the probability that migrants from any given population reach another site), and e is the local extinction rate of inhabited patches. Importantly, there is no mention of within-population dynamics. In this simplistic model, subpopulation size and within subpopulation reproduction and mortality rates are not considered; presence-absence data are sufficient to determine metapopulation structure, assuming that subpopulations are of equal size.

Levins' model has a few take-home points: (1) A metapopulation persists as long as individuals can successfully disperse from one patch to another and this migration rate exceeds the local extinction rate. (2) The pattern of patch occupancy by birds varies with time, but at equilibrium, the proportion of patch occupancy remains constant. The actual equilibrium proportion of occupied patches can be

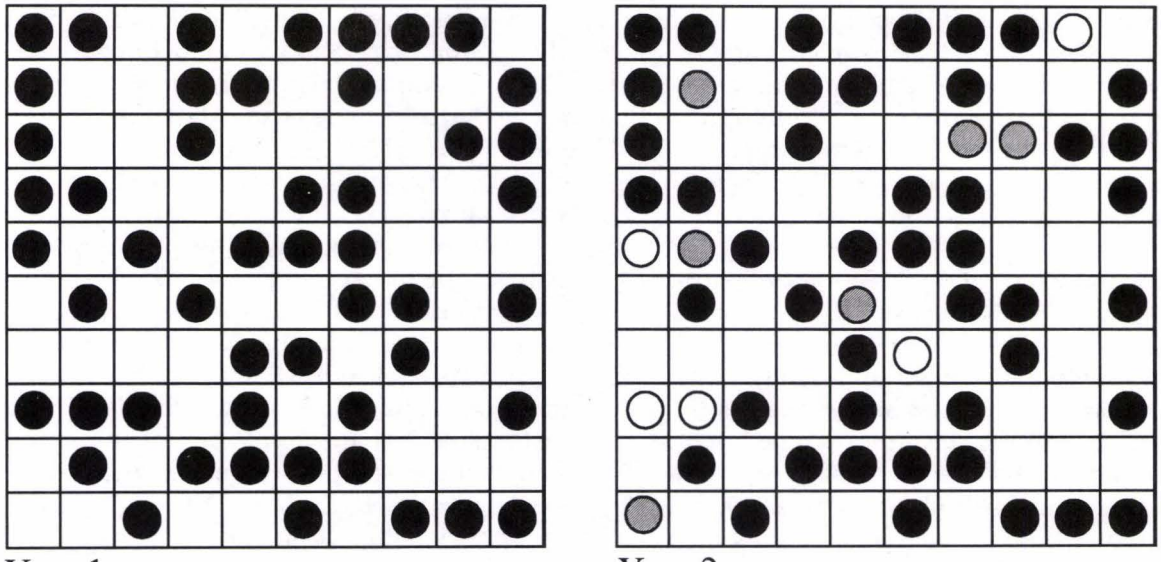

Figure 1.—*Hypothetical example of metapopulation dynamics for a species with 100 suitable habitat patches in two years. Darkened circles represent patches that are occupied in year 1 or year 2. Empty circles in year 2 represent local extinctions, and gray circles represent newly colonized habitat. In year 1 (left),* n_1 *= 50 or 50% of the suitable habitat is occupied, migration rate (m) = 0.225, and extinction rate (e) = 0.1 (estimates derived from Villard* et al. *1992). In year 2 (right),* n_2 *= 51 or 51% of the suitable habitat is occupied; five patches went extinct and six patches were colonized. The equilibrium population size for this metapopulation is 1 - e / m = 1 - (.1/.225) = .56 = 56% of suitable habitat occupied over time.*

large or small, and a metapopulation will persist as long as the migration rate exceeds the extinction rate.

It is fairly easy to see that few real populations satisfy the conditions of Levins' model. Classical (Levins') metapopulation structure has been documented for only a handful of species (Harrison 1994), and these species typically occur in early successional or ephemeral habitat and possess relatively poor dispersal capabilities. Some examples are frogs in temporary ponds (Sjogren 1991) and butterflies in disturbed habitats (Harrison *et al.* 1988, Thomas 1994). Most species, however, do not conform to Levins' assumptions because subpopulations invariably are nonrandomly configured in space and do not have uniform extinction and recolonization probabilities. Numerous spinoffs of Levins' model have been devised in an attempt to incorporate real-world processes. In general, these spinoffs still attempt to explain and predict the spatial distribution of occupied and unoccupied habitat patches.

For birds, metapopulation structure additionally might be obscured by the time scale at which extinction and recolonization occur. If extinction rates are low, for example, but dispersal between subpopulations is high, extinction-colonization episodes will be rare and metapopulation structure will be difficult to recognize. Bird species that appear to consistently occupy the same habitat patch over time might be overlooked as candidates for metapopulation structure since local extinction and local recolonization are necessary components of metapopulation structure. A typical field study may be too brief to detect "classical" metapopulation structure (sensu Levins 1970), even when subpopulation structure is present.

To summarize, metapopulation structure exists when a population of individuals is subdivided into semi-independent breeding populations that are linked by dispersal, but the dynamics of the greater population are driven by local extinction and recolonization that act at the larger spatial scale. Determination of metapopulation structure requires (1) knowledge of

the scale at which subpopulations are linked by dispersal, (2) knowledge of what constitutes "suitable" habitat, and (3) documentation of subpopulation presence or absence across suitable habitats over time. In addition, tests of classic metapopulation structure require that the additional assumption of uniform extinction and recolonization rates be met.

Empirical Evidence of Metapopulation Structure

In North America, evidence that neotropical migratory passerines exist as metapopulations is weak and has only recently become a major research topic (Villard *et al.* 1992). As far as we are aware, in North America, metapopulation structure has been considered only for forest-nesting passerines in highly fragmented landscapes (but see Opdam 1991). Hence, we will use forest-breeding migratory passerines to illustrate the evaluation of metapopulation structure.

First, many species of forest-nesting passerines occur as distinct, spatially separated subpopulations during the breeding season. Ovenbirds, for example, occur only in forests > 300 ha. (Hayden *et al.* 1985). Ovenbird territory size is generally a few hectares or less and ovenbirds hold breeding territories throughout the breeding season (Van Horne and Donovan 1994). Therefore, ovenbirds that populate a particular fragment are effectively isolated from breeding subpopulations in other fragments.

Second, recent studies document the disappearance and re-appearance of some forest-nesting passerines on a local scale (Brawn and Robinson 1996). Thus, local populations experience periodic extinction and recolonization, a condition of classical metapopulation models. In one example, bird populations have been censused since 1927 in Trelease Woods, an isolated woodlot in central Illinois (Kendeigh 1982). In most years, several breeding pairs of wood thrush occurred in the woodlot, but in other years, three extinction events were recorded followed by three colonization events (Brawn and Robinson 1996). During the time of wood thrush absence, Trelease Woods changed relatively little in its vegetation characteristics, and apparently suitable habitat remained unoccupied, another characteristic of metapopulations.

Finally, the only direct field test for metapopulation structure in forest nesting birds demonstrated that extinction and recolonization occurred and met conditions needed for metapopulation structure (Villard *et al.* 1992). Villard *et al.* (1992) studied three forest nesting species (ovenbird, wood thrush, and scarlet tanager) in 71 forest patches within an agricultural matrix near Ottawa, Ontario. Each of the forest patches was surveyed over a 2-year period, and each species was scored as present or absent in the two successive years. Occupied patches in year 1 that became unoccupied in year 2 constituted a local-scale extinction, and unoccupied patches in year 1 that became occupied in year 2 constituted a local-scale recolonization. Although patch occupancy shifted from year 1 to year 2, recolonizations exceeded or equaled local extinctions, a necessary outcome if the metapopulation is to persist (Villard *et al.* 1992). The authors concluded that the relevant demographic unit for these species in fragmented forests consists of a network of interacting patch populations.

Although these studies indicate that some aspects of metapopulation structure are exhibited by North American songbirds, none provide sufficient evidence that populations are structured as metapopulations and that the overall persistence of a species is dictated by a balance of extinction and recolonization of habitat patches. However, these studies force managers to consider patch dynamics for bird species, and metapopulation theory provides impetus for documenting dispersal patterns, the scale at which subpopulations are linked, and the factors that promote or retard dispersal (Villard *et al.* 1995).

Management Implications for Metapopulation Dynamics

Managers must consider several critical issues when evaluating whether metapopulation dynamics pertain to a given bird species, including identifying the effective scale and boundaries of the metapopulation, the definition of subpopulations or suitable habitat patches, and rates of dispersal and local patch occupancy over time (Harrison 1994). First, in defining the boundaries of a metapopulation, managers need to consider the scale at which bird subpopulations interact by dispersal; this scale defines an appropriate demographic unit

for management. For example, if dispersal occurs within a Breeding Bird Survey physiographic region (defined by history, vegetation, and soil characteristics), then all suitable habitat patches should be evaluated in terms of presence and absence within that region to evaluate possible metapopulation structure. Focusing on a smaller region than the actual scale of dispersal could result in misinterpretation of presence and absence patterns and the status of the metapopulation.

Second, once this scale of dispersal is defined, managers should consider the spatial distribution of priority species (its presence and absence in space across time). This includes considering how interbreeding individuals are distributed across habitats to determine what constitutes a subpopulation. For example, ovenbirds in the Midwest that inhabit relatively large (>100 ha) forest patches collectively represent isolated subpopulations during the breeding season; dispersal among patches in successive years occurs when birds disperse from one breeding patch to another or from birth location to the first breeding territory (Martin and Li 1992). By contrast, Baltimore orioles that inhabit relatively small (<10 ha) habitat patches often move from patch to patch within a breeding season (Howe 1979, 1984), and thus an isolated breeding subpopulation may encompass an archipelago of small habitat islands (fig. 2) (Rolstad 1991, Weins 1989). Thus, the definition of a subpopulation in a metapopulation context is likely to be species specific and to depend on species-specific habitat requirements.

Although we do not know the scale of dispersal in most species and are challenged to define what constitutes a subpopulation, metapopulation theory promotes active management at spatial scales larger than the traditionally viewed local scale. One consequence of dispersal among subpopulations is that managers should realize that actions taken at their particular location can influence the dynamics of bird populations some distance from their location. For example, suppose a large forest patch is managed to the detriment of a species. If this large patch is a significant part of the metapopulations habitat, declining habitat quality in this large patch may put subpopulations in other patches at risk as well. In some instances, this could even result in the extinction of the metapopulation. Furthermore, management actions at the local scale contribute to the extinction and recolonization rates of the larger metapopulation and thus can influence population dynamics beyond the local scale. Likewise, management actions that promote the presence of a species at the local scale may be offset by management actions elsewhere that impact the number of colonists available to disperse to their patch (Lande 1991).

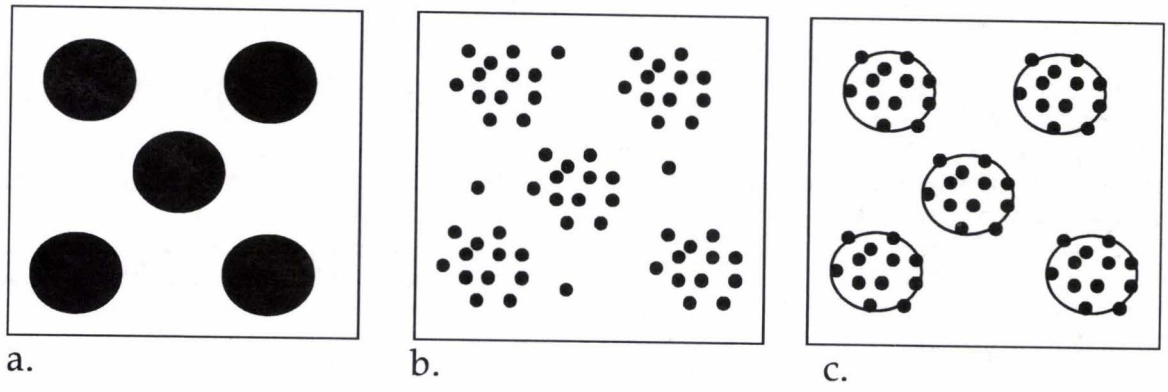

Figure 2.—*Hypothetical example of subpopulation identification: (a) darkened circles represent habitat patches with each patch containing demographically independent subpopulations, (b) darkened circles represent a patchy habitat mosaic over which all individuals move and exchange genes, therefore one population (or subpopulation is depicted); (c) darkened circles represent a patchy habitat mosaic, and although individuals move across habitat patches, demographically independent subpopulations exist because gene exchange is limited within open circles.*

From a management standpoint, the extinction of a subpopulation from apparently suitable habitat is a normal occurrence in metapopulation dynamics. Metapopulation equilibrium focuses on the proportion of patches that are occupied over time. Complete loss of one subpopulation may or may not influence this equilibrium proportion. But this does not imply that local extinctions should be considered lightly (Harrison 1994). If suitable habitat becomes unsuitable, the total number of patches available to the metapopulation is reduced. If equal population size in all patches is assumed, the result is a decline in the total number of individuals in the metapopulation, which in turn influences the pool of migrants that disperse from patch to patch.

SOURCE-SINK DYNAMICS: CONCEPTS AND EMPIRICAL EVIDENCE

Source-Sink Concepts

Resource managers have long recognized that reproductive and mortality rates can vary tremendously across a species range, and there is evidence of this for midwestern neotropical migratory birds (e.g., Robinson *et al.* 1995). Source-sink models were created to describe organisms in spatially discrete and demographically independent subpopulations, similar to the metapopulation models described above. In this situation, however, the key demographic question is how a greater population can persist at equilibrium when its subpopulations vary in their birth and death rates (Pulliam 1988 and references therein). Thus, where metapopulation models assume that subpopulations have equal probability of extinction and recolonization, source-sink models recognize that some subpopulations (sinks) are inviable without constant influx of immigrants from other subpopulations, while others (sources) remain viable without the influx of any immigrants.

Pulliam used "BIDE" models (Cohen 1969, 1971) to determine how a greater population that is comprised of subpopulations varies in number over time. In a BIDE model, B, I, D, and E represent birth, immigration, death, and emigration, respectively. All subpopulations contribute to the equilibrium number. As stated, each subpopulation is an independent demographic unit and has its own bide parameters. A subpopulation is in dynamic equilibrium (not changing) when b + i - d - e = 0. At equilibrium, a subpopulation is a source when b > d and e > i, and is a sink when b < d and e < i. The greater population is at dynamic equilibrium (not changing) when b (all the births) + i (all the immigrants from outside the greater population) - d (all the deaths) - e (all the emigrants that leave the greater population) = 0.

For example, consider a very simplistic global population at equilibrium that consists of 100 individuals, distributed across four discrete patches (fig. 3). Each subpopulation varies in its birth, death, emigration, and immigration rates. The bide parameters represent per capita rates and thus are not area specific. One habitat is a source, with an initial population of 10 breeding adults (5 pairs), where b = 2.5 young per year, d = 0.5 deaths per year, i = 0, and e = 2.0 emigrants per year. (Thus, b > d, and e > i, but the population overall does not change over time). Three habitats are sinks, where the initial number of individuals is 40, 25, and 25, and in which b = 0.25, 0.4, 0.52, d = 0.5, 0.52, 0.8, and i = 0.25, 0.12, 0.28, and e = 0, 0, 0 respectively. Thus, in all sinks, b < d and e < i, but the population numbers of each sink does not change over time. Each subpopulation is in equilibrium, and the overall (global) population is also in equilibrium (b = 0.58, d = 0.58, i = 0, and e = 0).

This example depicts a population at equilibrium (N = 100 individuals). In general, Pulliam considered a species that occurs in both source and sink habitat in which the population size in the source is regulated by some resource that cannot be infinitely subdivided between individuals (e.g., the number of possible breeding territories or nest-hole cavities). Individuals unable to find a breeding site in the source emigrate to the sink because a poor breeding site is better than none at all. If there are many habitats, the total population reaches an equilibrium when the total surplus in all the source habitats equals the total deficit in all the sink habitats.

Some basic take-home points from Pulliam's source-sink model are: (1) At equilibrium, the **number of individuals** in the overall, global population is not changing. This is quite different from the equilibrium point in Levins' metapopulation model, which focuses on the **proportion of occupied patches** over time

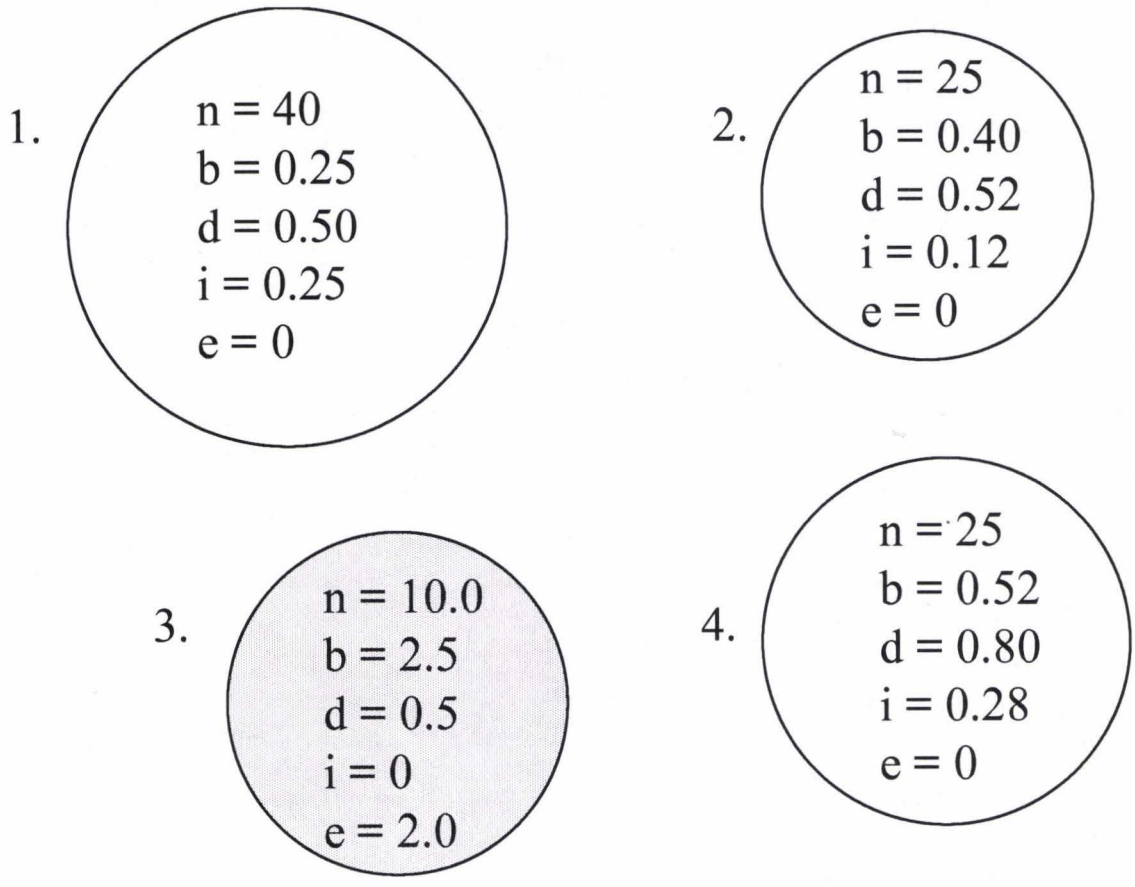

Figure 3.—*Hypothetical example of a source-sink model with four demographically independent subpopulations, three sinks and one source. The number of individuals (n), birth rate (b), death rate (d), immigration rate (i), and emigration rate (e) are given for each subpopulation. Each subpopulation is at equilibrium (growth = 0), and the overall population is not changing. Source: 2.5+0-(2+.5) = 0; Sink 1: .25+.25-(.5+0)=0; Sink 2: .4+.12-(.52+0)=0; Sink 3: .52+.28-(.80+0)=0.*

(although if subpopulations are assumed to be of equal size, then the number of individuals at equilibrium in a metapopulation will remain unchanged, but the distribution of individuals is constantly shifting); (2) within-subpopulation dynamics are important in determining the overall equilibrium population size, since the numbers of individuals on each patch and their growth rates are implicit in the model. Source and sink subpopulations can be characterized by their "strength," depending on their intrinsic rate of growth and the number of individuals present; (3) the source-sink status of a subpopulation has nothing to do with the size (number of individuals) within the subpopulation. Sinks can support a vast number of individuals and sources can be numerically very small. Nonetheless, sources must have enough individuals with a high enough per capita production to support sink populations.

A variation in Pulliam's source-sink model was introduced by Howe *et al.* (1991) and Davis and Howe (1992), who added a density-independent dispersal term to the source-sink system. In their model, emigrants (e.g., juveniles) leave a subpopulation even when unoccupied territories might be available. Population simulations show that changes in the conditions of local subpopulations (e.g., as a result of logging) can have significant impacts on other subpopulations, even if the affected subpopulations are not demographic sources (see also Donovan *et al.* 1995b).

Empirical Evidence of Source-Sink Dynamics

Many field studies show that reproductive success of neotropical migrant birds varies across a species' range (e.g., Probst and Hayes 1987, Robinson *et al.* 1995), but few examine the interaction of subpopulations from a source-sink viewpoint. To evaluate source-sink dynamics, one must know the BIDE parameters of each subpopulation. To identify source or sink habitat, the fecundity and death rates of adults in the subpopulation must be measured. Annual censuses, therefore, are not capable of establishing source-sink status unless local abundance is strictly correlated with productivity (Brawn and Robinson 1996).

Most empirical studies that document sink populations use nesting and mortality data from the subpopulation and model population persistence over time in the absence of immigration or emigration (e.g., Ricklefs 1970, Stacey and Taper 1992, Pulliam and Danielson 1991, Donovan *et al.* 1995a). Without immigration, sink populations decline over time and will eventually go extinct. Returning to our forest bird examples, many forest bird populations in Illinois woodlots suffer high rates of predation by mammals/birds and parasitism by brown-headed cowbirds and, thus, do not produce enough young to balance adult mortality (Robinson 1992, Brawn and Robinson 1996). Similarly, demographic models of three forest species on forest fragments in Wisconsin and Missouri show that, in the absence of immigration, populations would decline over time to extinction (Temple and Cary 1988, Donovan *et al.* 1995a).

The same demographic characteristics (fecundity and mortality rates) are used to identify source habitats. Sources, by definition, have more births than deaths, but equilibrium is maintained because emigration is greater than immigration. Most empirical studies document "potential" sources, and use nesting data and mortality data from the subpopulation to show that populations would either stay the same or grow over time in the absence of immigration (Ricklefs 1970, Sherry and Holmes 1992, Donovan *et al.* 1995a). If it is shown that reproductive success is very high relative to mortality, and yet population size remains nearly constant over time, then source status can be inferred because emigration is necessary for the population to be stable in size.

Management Implications of Source-Sink Demography

Source-sink demography provides additional implications for population management. Like metapopulation dynamics, independent breeding subpopulations are linked by dispersal and the scale at which sources and sinks interact is unknown (Brawn and Robinson 1996). In the absence of dispersal data, regional-scale management is preferred over local-scale management (Robinson *et al.* 1995). Furthermore, local-scale management almost certainly will have regional-scale impacts. Since dispersal in birds probably does not follow political boundaries, we suggest a physiographic or regional approach until more information is obtained.

Source-sink dynamics might operate at many scales, and demographic units might vary in size over time and space. Sinks depend on sources for persistence. A single source patch may well be able to support several sink patches, which may be larger numerically than the source itself. If that source is extinguished through habitat destruction, a catastrophic event, or a change in management policy, all the sinks that it supports (assuming one source supplies many sinks) inevitably will become extinct as well. Thus, the complete or even partial loss of a source subpopulation directly affects the long-term viability of a spatially subdivided population or population mosaic. Sources that produce a large number of young per unit area should be identified because they contribute a relatively large number of individuals to the larger population. Sources that produce a small number of young per unit area may be of less management concern, but nevertheless they may be important in maintaining regional populations. For example, they may serve as stepping stones that promote dispersal to other patches. They are also potential refuges from local catastrophes that may befall other patches. As in metapopulation systems, managers should realize that actions taken at one location can influence population dynamics in another location (Temple and Cary 1988, Howe *et al.* 1991, Donovan *et al.* 1995b).

Additionally, the habitat associated with source or sink status varies with species. Since managers often must manage for multiple species that may have conflicting habitat needs, increasing source habitat for one species may mean decreasing source habitat for another. Thus, managers need to assess the source-sink status of species of interest and determine how actions taken on a local scale can affect their local population as well as distant subpopulations that may depend on them.

It may be tempting for managers to eliminate a patch of sink habitat since it cannot support demographically viable populations. However, because bird populations are linked, maintenance of sink habitats can be very important. First, the bulk of a population may reside in a sink at any point in time (Pulliam 1988, Van Horne 1983, Howe *et al.* 1991, Donovan *et al.* 1995b). Sources, although demographically viable, may be subject to stochastic extinction risks, and sink habitat can bridge source populations that periodically go extinct. Furthermore, sinks may be the genetic storehouse of unique or rare genes or may facilitate gene flow between sources. Individuals in sinks may contribute significantly to the genetic diversity of the larger population if individuals in the sink can successfully reproduce (Allendorf 1983). However, if sinks act as ecological traps that draw birds away from higher quality source habitats (Gates and Gysel 1978, Pulliam and Danielson 1991) or act as a conduit to spread risk or disease, maintenance of sink habitats may not be desirable. These factors should be weighed in any management action.

SUMMARY OF MANAGEMENT IMPLICATIONS

The practical implications of traditional metapopulation models and source-sink models are similar. In both cases, dispersal between local areas plays a central role in population dynamics. Field studies of neotropical migrant birds provide evidence (albeit indirect) that isolated breeding populations are linked by dispersal. The scale at which populations interact by dispersal is largely unknown; understanding this scale should be a primary research objective in the future. In the absence of dispersal data, regional-scale management (of physiographic regions or greater) is preferred over traditional, local-scale management.

Both metapopulation models and source sink models emphasize the need to set management goals at the regional scale. Because of dispersal from one location to another, setting management goals directed solely at the local scale may have limited success because populations may be regulated by events occurring at scales beyond the local scale. For example, management actions directed at decreasing the abundance of a species (such as cowbirds) at the local scale may be futile if ample cowbird habitat is available beyond the local scale and allows cowbirds to infiltrate the local habitat of concern. Similarly, management actions directed at increasing a priority species at the local scale may be offset by management actions elsewhere that impact the number of birds available to disperse to their patch. Thus, management goals should be targeted at the regional scale, and local managers should realize that their actions may impact dynamics at the local scale and beyond.

We conclude by providing some general management considerations that pertain to both metapopulation and source-sink structure in birds.

Metapopulation considerations:

(1) In a metapopulation context, managers should realize that presence or absence of a priority species in their locations may be part of a larger process that involves subpopulations beyond their managed areas.

(2) Since local extinction and local recolonization rates are key parts of the metapopulation paradigm, managers should promote conditions that decrease local extinction or increase the number of colonists (typically young birds) that can disperse to other areas.

(3) Local extinction is a natural part of metapopulation processes, but decreasing habitat availability for a priority species changes the total number of habitat patches available for the priority species; loss of suitable habitat patches potentially decreases the population size. Therefore, habitat should be maintained for priority species.

Source-sink considerations:

(1) Each priority species should be evaluated separately to determine habitats associated with high and low productivity. Because priority species often have conflicting habitat needs, increasing the amount of source habitat for one species may decrease source habitat for a second species.

(2) Identifying and maintaining source populations for priority species is vital for the persistence of regional populations. Annual variation in quality can change the source-sink status of a particular habitat patch. Density and local population trends are not useful in establishing source-sink status; sinks can be numerically large and show no sign of population decline, and sources can be numerically small and show population decline due to emigration.

(3) Not all sources are equal; a source that produces a high number of young per unit area may be of more value to the larger population than a source that produces a low number of young per unit area; thus assessment of priority species should include evaluation of where each management area is in relation to the center of distribution of high priority species.

(4) Sink habitat may house many individuals. Even if reproductive success is low in a sink, a large number of young can still be produced there depending on the number of adults. However, if birds readily disperse away from high-quality source habitats to low-quality habitats, then sinks may be 'traps' that change the number of individuals in sources and sinks and thus can impact how many young are produced in a given year.

So far we have not considered how subpopulations are configured with respect to one another. An extension of the metapopulation/source-sink concept examines the specific distribution of individuals in space. These so called "spatially explicit" models can define populations as collections of individuals or collections of local subpopulations (Conroy *et al.* 1995, Dunning *et al.* 1995). Analysis of spatially explicit models inevitably involve computer simulations, and they require empirical information about the distribution of individuals in a given region. Although a detailed analysis of spatially explicit models is beyond the scope of this review, managers should be aware that such models provide tools for examining specific populations in appropriately large management units.

We have much to learn about the dynamics of heterogeneous landscapes and their influence on bird populations. Metapopulation and source-sink theories and their extensions provide a theoretical framework for understanding population dynamics at large and realistic scales. The critical need today is for a combination of improved theories and empirical information about specific neotropical migratory bird populations in specific management landscapes.

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MANAGEMENT OF NORTHERN PRAIRIES AND WETLANDS FOR THE CONSERVATION OF NEOTROPICAL MIGRATORY BIRDS

Douglas H. Johnson[1]

ABSTRACT.—Grasslands and wetlands of the northern prairies provide important breeding habitat for a number of birds. Deciding which species deserve most attention in managing those habitats depends, in part, on the importance of the area to the species. Many species in northern prairies are more common elsewhere and need no special consideration in that area. Several species, however, are critically dependent on the prairies. These species merit particular attention if protection of biodiversity is a goal.

Both grasslands and wetlands in the northern prairies have been extensively converted for agricultural use, which has reduced the value of these habitats for breeding birds. Most land-use changes took place before monitoring programs for birds began, so quantitative assessments of changes in avian populations are lacking. This paper discusses the status of bird populations in the northern prairies, key upland and wetland habitats, effects of common management practices, and issues that specifically result from a landscape perspective. Most management practices are employed for other objectives; consequences to nongame birds are incidental, but vitally important to some species.

The northern prairies constitute a major breeding area for many wetland- and grassland-dependent species of birds. I review the status of bird populations in the northern prairies, key upland and wetland habitats, effects of common management practices, and issues that specifically arise from a landscape perspective. I focus on the United States portion of the northern tallgrass and mixed-grass prairies, including northern Iowa, western Minnesota, portions of North Dakota and South Dakota east of the Missouri River, and northeastern Montana. Certain conclusions will be more widely applicable, especially to the southern prairie provinces of Canada.

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[1] *Northern Prairie Science Center, National Biological Service, Jamestown, ND 58401.*

BIRD POPULATIONS OF THE NORTHERN PRAIRIES

A large number of bird species breed in the northern plains. The avifauna includes species of boreal, eastern, southern, and western affinities (Stewart 1975, Johnsgard 1979). Most species are more common elsewhere; I emphasize species for which the area is important because it supports a significant proportion of the species' population. I concentrate mostly on neotropical migrant landbirds, although some short-distance migrants are also included for completeness and comparison (table 1).

Stewart (1975) classified the breeding birds of North Dakota according to their biogeographical affinities (table 2). Of the 190 species included, 56 (29 percent) were associated with the north-central, mixed-grass avifauna. Those 56 species made up 80 percent of the total breeding bird population in 1967 (excluding exotic species).

Table 1.—*Migratory status*[a] *and population trends*[b] *(based on BBS results in the Central Region) of birds of the northern prairies.*

Species	Trend 1966-1994	Trend 1966-1979	Trend 1980-1994
PRIMARILY NEOTROPICAL MIGRANTS			
Swainson's hawk	0.9	1.3	1.6
Upland sandpiper	2.1 ↑↑↑	3.7 ↑↑↑	0.3
Burrowing owl	-2.8	2.3	-0.5
Willow flycatcher	NA	NA	NA
Western kingbird	1.1 ↑↑↑	1.0	-0.0
Eastern kingbird	0.2	0.4	0.4
Common yellowthroat	-0.9 ↓↓↓	1.8 ↑↑↑	-2.1 ↓↓↓
Dickcissel	-1.0 ↓↓↓	-4.5 ↓↓↓	1.3 ↑↑↑
Clay-colored sparrow	-1.1	-0.9	0.9
Lark bunting	-0.7	-4.0 ↓↓↓	1.1
Baird's sparrow	-0.9	-4.0 ↓	-0.5
Grasshopper sparrow	-2.9 ↓↓↓	-2.6 ↓↓↓	-1.8 ↓↓
Bobolink	-2.4 ↓↓↓	-3.2 ↓	-3.0 ↓↓↓
Yellow-headed blackbird	0.5	3.3	-2.1 ↓↓
PRIMARILY TEMPERATE MIGRANTS			
Northern harrier	-2.1 ↓↓	-1.9	-0.3
Ferruginous hawk	6.2 ↑↑	5.5	11.6 ↑↑↑
Killdeer	-0.3	3.0 ↑↑↑	-2.0 ↓↓↓
Willet	-1.8	4.7 ↑↑↑	-0.4
Marbled godwit	0.7	7.9 ↑↑	—
Short-eared owl	2.0	27.7 ↑	-0.5
Horned lark	-0.6	-0.2	-1.1 ↓↓↓
Sedge wren	1.3	-4.1 ↓↓	5.7 ↑↑
Marsh wren	3.6	-4.9 ↓↓	6.7 ↑↑↑
Gray catbird	-1.0	0.7	-0.8
Sprague's pipit	-0.1	-6.5 ↓↓↓	3.0
Loggerhead shrike	-3.1 ↓↓↓	-4.3 ↓↓↓	-1.3 ↓↓
Vesper sparrow	0.4	0.3	1.1 ↓
Lark sparrow	-3.8 ↓↓↓	-5.6 ↓↓↓	-2.8 ↓↓↓
Savannah sparrow	0.5	-2.0	1.5
Song sparrow	1.4 ↑↑↑	-0.2	2.8 ↑↑
Swamp sparrow	1.3	6.3 ↑↑↑	2.5
McCown's longspur	3.7	11.0 ↑↑	5.8
Chestnut-col. longspur	-0.4	0.5	-0.6
Red-winged blackbird	-0.5 ↓↓	1.1 ↑↑↑	-1.3 ↓↓↓
Western meadowlark	-0.3	-1.1 ↓	0.3
Brown-headed cowbird	-0.5 ↓	2.5 ↑↑↑	-0.2
PRIMARILY PERMANENT RESIDENTS			
Greater prairie-chicken	-0.9	16.8 ↑↑↑	5.4
Sharp-tailed grouse	3.8 ↑	2.7	8.0

[a] Based primarily on Gauthreaux (1992).

[b] Estimated annual percentage change;

↑ increasing at $P < 0.10$; ↑↑ increasing at $P < 0.05$; ↑↑↑ increasing at $P < 0.01$; ↓ decreasing at $P < 0.10$; ↓↓ decreasing at $P < 0.05$; ↓↓↓ decreasing at $P < 0.01$.

Table 2.—*Faunistic composition and total population composition of breeding birds in North Dakota (Stewart 1975).*

Avifauna	Composition by species	Composition by population
North-central North America (mixed-grass)	56 (29%)	80%
Eastern North America[1] (mostly woodland)	71 (37%)	13%
Western North America[1]	37 (19%)	6%
Northern North America	31 (16%)	1%

[1] Five species are included in both eastern and western avifaunas.

I suggest species deserve special attention in the northern prairies if a significant portion of their population breeds in the area and they meet any of the following other criteria: (1) their breeding range is small, (2) their total (continental) population is small, (3) they have declined in number or contracted in geographic range, (4) they are restricted to a narrow range of habitats, especially if those habitats are threatened, or (5) there is some major potential threat to their population. The most compelling reason for emphasizing any particular species in an area is that the area supports a substantial portion of the continental population of the species. That point may seem obvious, but considerable management attention is directed toward species in insignificant portions of their range (as noted, for example, by Knopf 1992). This effort may be appropriate if such peripheral populations are genetically distinct from central populations and offer greater potential for adaptation to changing environments (Lesica and Allendorf 1995), but that situation is unlikely to hold for widely dispersing migratory birds. With this perspective, the scheme used by Partners In Flight provides a prioritization scheme for landbirds, based on perceived threat of extinction (Hunter *et al.* 1993). Alternatively, a focus on endemic species (e.g., Knopf 1988) is valuable but may miss some species that are not endemic but in need of attention.

By such criteria, species such as Baird's sparrow and Sprague's pipit, which have small populations and whose breeding ranges are restricted to the northern Great Plains, deserve more attention in that area than do species such as the brown thrasher and yellow warbler. The latter species also breed in the northern Great Plains, but their distributions are far broader, and they are more common elsewhere. Even in the Plains, they can use artificial habitats such as shelterbelts and suburban plantings, which are increasingly common. Conversely, the Baird's sparrow and Sprague's pipit require grassland, the natural habitat on the northern Plains.

Status and Trends

The mid to late 1960's is a convenient reference point for the status of bird populations in this region. The Breeding Bird Survey (BBS) began in 1966 and became operational in the region in 1967. Also, a statewide survey of North Dakota birds was conducted in 1967 (Stewart and Kantrud 1972); a repeat of that survey in 1992 and 1993 provides a useful contrast (Igl and Johnson 1995b). Considering population changes during the last 25 years or so can be misleading, however. Most of the major changes in habitat in the northern prairies occurred after settlement by Europeans but before the 1960's, and associated changes in bird populations were not tracked by BBS or other programs. Our knowledge of bird populations prior to European settlement is weak, based on comments by early explorers and settlers or inferred from current bird use of habitats that have not been altered dramatically.

Early reports mention huge numbers of waterfowl, shorebirds, and other birds (e.g., Dinsmore 1994). The reports lack quantification, and it is questionable whether low numbers or absences would be reported as faithfully as extreme abundances. Nonetheless, many of the accounts describe grassland birds in numbers unheard of today.

BBS results indicate that during 1966-1991, grassland-nesting birds had a higher proportion of declining species than did any other avian guild in North America (Droege and Sauer 1994, Knopf 1994). BBS trends for the 1966-1994 period are given in table 1 for the Central Region, roughly the area between the Rocky Mountains and the Mississippi River.

From 1966 to 1994, significant decreases outnumbered significant increases by four species to two among neotropical migrants, and by five species to two among temperate migrants (table 1). During the more recent period (1980-1994), significant declines outnumbered significant increases by four to one for neotropical migrants and matched them at five to five among temperate (short-distance) migrants. Declines were consistent in both early (1966-1979) and late (1980-1994) periods for only the grasshopper sparrow, bobolink, and Baird's sparrow, among neotropical migrants, and for northern harrier, horned lark, loggerhead shrike, and lark sparrow, among temperate migrants. Increases were consistent in both time periods for the neotropical migrants, Swainson's hawk, upland sandpiper, and eastern kingbird; for the temperate migrants, ferruginous hawk, vesper sparrow, swamp sparrow, and McCown's longspur; and for the permanent resident sharp-tailed grouse.

Populations of temperate migrants in North Dakota in 1992-1993 did not differ consistently from those in 1967 (Igl and Johnson 1995b; table 3). Numbers of long-distance migrants increased, however, and those of permanent residents more than doubled from the early to the recent period. Examining bird populations by primary breeding habitat, Igl and Johnson (1995b) concluded that species that rely on trees (open habitat with trees, woodland and woodland-edge, and residential-generalist) consistently increased from 1967 to 1992-1993 (table 3). Trends for groups of species associated with other habitat types were not evident.

Such groupings, however, can disguise changes occurring to particular species. Among the grassland birds, numbers of chestnut-collared longspurs, western meadowlarks, savannah sparrows, and Baird's sparrows declined by 39 percent or more; clay-colored sparrows and bobolinks declined at lesser rates (table 4). Horned lark and lark bunting numbers varied without a trend, likely due to changes in precipitation during the study years. Counts of vesper sparrows and upland sandpipers increased by more than 50 percent.

Table 3.—*Mean number of indicated breeding pairs in 128 randomly selected quarter-sections in North Dakota by year, migration strategy, and preferred breeding habitat.*

	Mean pairs/100 ha		
Migration strategy	**1967**	**1992**	**1993**
Permanent resident	2.6	5.7	6.1
Short-distance migrant	95.5	74.7	99.5
Long-distance migrant	43.2	52.3	45.4
Breeding habitat			
Wetland/wet meadow	37.5	24.7	32.6
Grassland/open habitat	71.7	59.3	68.3
Open habitat with trees	5.5	10.6	9.7
Shrubland	7.2	7.5	9.0
Woodland/woodland-edge	15.6	24.6	25.3
Residential/generalist	3.7	5.7	5.8
Other	0.1	0.3	0.3
Total	141.3	132.7	151.0

Table 4.—*Number of indicated pairs of the 20 most common grassland bird species observed on 128 randomly selected quarter-sections in North Dakota in 1967, 1992, and 1993.*

Species	Number of indicated pairs		
	1967	1992	1993
Horned lark	1,253	1,093	1,661
Chestnut-collared longspur	1,129	602	755
Red-winged blackbird	945	597	710
Western meadowlark	926	487	646
Brown-headed cowbird	460	643	610
Lark bunting	604	679	298
Grasshopper sparrow	301	402	449
Mourning dove	292	339	337
Savannah sparrow	516	134	276
Clay-colored sparrow	364	261	289
Vesper sparrow	195	224	393
Eastern kingbird	167	321	245
Bobolink	216	186	172
Western kingbird	103	194	177
Common yellowthroat	134	91	175
American goldfinch	106	146	132
Baird's sparrow	170	77	125
Killdeer	105	112	142
Upland sandpiper	63	106	89
Wilson's phalarope	73	30	36

HABITATS OF THE NORTHERN PRAIRIES

The primary natural habitat type in the northern plains is grassland. Three broad provinces of grassland in the Great Plains are generally recognized, which correspond to a gradient of increasing precipitation from west to east: shortgrass prairie in the west, mixed-grass prairie in the center, and tallgrass prairie in the east (Risser *et al.* 1978). Patches of one grassland type can be found within another province, depending on local edaphic features, topography, precipitation patterns, and land use. The focus here will be on the mixed-grass and tallgrass prairies of the northern United States.

Prior to settlement by Europeans, the northern plains were a vast grassland; trees were scarce or absent (e.g., Bragg and Steuter 1995). Early reports indicate that trees were largely restricted to river floodplains, east- or north-facing bluffs along streams, and prominent hillsides (Stewart 1975:4, Bragg and Steuter 1995). Grasslands were maintained by periodic drought; fires, especially to the east; and, especially in the west, grazing by large herds of herbivores such as bison. These forces created mosaics of habitat ranging from heavily grazed to undisturbed (England and DeVos 1969).

In the eastern portion of the northern plains, innumerable depressions were left when the Wisconsin glacier retreated about 10,000 years ago. These wetland basins, called prairie potholes, contain water for various lengths of time in most years (Stewart and Kantrud 1971). The most ephemeral wetlands may hold spring runoff or summer rains for only a few days. At the other extreme are lakes, which almost never go dry. In between are seasonal wetlands, which in a typical year contain water from early spring until mid to late summer, and semipermanent wetlands, which in most years are wet throughout the frost-free season. Less common are alkali wetlands—large, shallow basins with such high alkalinity that salts are blown out when the wetland is dry, and where no emergent plants grow when it is wet. Another unusual wetland type is the fen, characterized by

floating or quaking mats of vegetation caused by groundwater seepage. Different wetland types support different kinds of vegetation and, in turn, different animal communities.

Critical to understanding the prairie is recognizing its dynamic nature, particularly as driven by recurring droughts. Prairie occurs primarily under semi-arid conditions. Precipitation is generally inadequate for growth of most woody vegetation, and the herbaceous vegetation favored fires and supported large herds of grazing herbivores, both features that further discouraged woody growth. Drought is essential to wetlands as well as uplands. The periodic drying of wetland basins facilitates nutrient cycling and results in high productivity when water returns (Murkin 1989).

Changes in Habitats of the Northern Prairies

Much of the terrestrial grassland habitat has been cultivated for crops. This conversion is nearly total in the eastern portion; tallgrass prairie is one of the most threatened habitats in the northern plains, with only scattered fragments remaining (Samson and Knopf 1994, Noss *et al.* 1995). Less mixed-grass prairie has been cultivated, largely because the terrain is rougher and precipitation is lower and less predictable. Irrigation has in many places rendered lands more suitable to cultivation, however. More shortgrass prairie remains, although much of it is intensively grazed by domestic livestock. Small grains such as wheat, barley, and oats are common crops in the western plains; in the east, row crops such as corn, soybeans, sunflowers, and potatoes are also planted.

Settlement of the northern plains by Europeans brought major increases of woodland. Tree claims were planted to protect farmsteads from the ever-present winds, and shelterbelts were established along field borders to reduce soil erosion, especially after the drought of the 1930's. Also, inadvertent increases of woody vegetation resulted from fire suppression by settlers (McNicholl 1988).

Prairie wetlands likewise have been altered in a number of ways. Drainage of basins to facilitate cultivation was very common, especially in the eastern prairies. Sometimes several small wetlands were drained into a larger one, which eliminated the smaller wetlands and altered the hydrology of the receiving wetland. Losses of wetland from settlement to 1980 were 27 percent in Montana, 35 percent in South Dakota, 49 percent in North Dakota, and 42 percent in Minnesota (Dahl 1990). Smaller, more temporary wetlands were more susceptible to drainage than were the larger, more permanent basins. Losses of some wetlands were partially offset by the creation of others. Stock-watering dams and dugouts have been constructed in the northern prairies, usually along intermittent streams. Several mainstem dams on rivers have created large reservoirs, although their value to breeding birds is limited.

Integration of Upland and Wetland Habitats

A landscape perspective requires consideration of broader-scale issues than does a local perspective. Diversity in a regional sense is more important than local species diversity (Knopf and Samson 1994). Maximizing the species richness of an area (species packing) is not a goal; maintaining viable populations is (Johnson *et al.* 1994b). This requires an understanding of each species' habitat needs and how different habitats relate to one another.

Species that forage in one habitat but nest in another illustrate connections between different habitats. Dabbling ducks feed in wetlands but commonly nest in upland grassland. Certain shorebirds, such as Wilson's phalarope, willet, and marbled godwit, likewise require both terrestrial and aquatic habitats. American bitterns and northern harriers will nest either in emergent wetland vegetation or in dense upland vegetation. Red-winged and yellow-headed blackbirds nest in wetlands but often forage in terrestrial habitats. Certain species take advantage of the dynamic nature of the prairies, settling in wetland cover as available, but using normally terrestrial vegetation during unusually wet periods. For example, the Le Conte's sparrow, which usually nests in wet swales, will nest in high numbers in upland grass-forb plantings during wet periods (Igl and Johnson 1995a).

Not any patch of habitat, even preferred habitat, will suffice; size of the patch may be influential. Several grassland species are area sensitive. For example, Herkert (1994a) found

that transects in larger grassland blocks were more likely to contain grasshopper sparrows, bobolinks, and savannah sparrows than were comparable transects in smaller blocks. Studies of birds in Conservation Reserve Program fields have indicated that many of the larger-bodied species such as northern harrier, short-eared owl, Wilson's phalarope, marbled godwit, and willet rarely occur in small habitat blocks (D. H. Johnson, in prep.). Marsh size and isolation influenced occupancy by wetland birds in Iowa (Brown and Dinsmore 1986). Even habitats that are used may not be effective in maintaining viable populations. Habitat patches may consistently attract breeding birds that fail to reproduce (sink habitats: Pulliam [1988]). Such areas occur in the northern plains for waterfowl (Klett *et al.* 1988, Greenwood *et al.* 1995). Although less is known about the population dynamics of passerines, cultivated fields (Rodenhouse and Best 1983) and especially hayfields (Bollinger *et al.* 1990, Frawley and Best 1991) likely operate as sink habitats.

Features in a landscape may affect bird use of habitats at some distances. Occupancy of grassland habitat can be influenced by the nearby presence of woody vegetation; Johnson and Temple (1986) reported nest densities of grasshopper sparrows were lower near trees than farther away, whereas the reverse held for clay-colored sparrows and western meadowlarks. Trees also may provide perch sites from which raptors can hunt and brown-headed cowbirds can seek host nests in which to lay their eggs. Johnson and Temple (1986) found that nest success of several grassland bird species was significantly higher for nests located far from a field-forest edge. Burger *et al.* (1994) reported similar results for artificial nests in grassland. Birds that nest in wetlands, woody areas, or human developments often forage in nearby grasslands, and may compete with grassland-dependent birds.

EFFECTS OF HABITAT MANAGEMENT

Publicly owned native grasslands are managed in several ways, but grazing and prescribed burning are the two most common active practices. Many public grasslands are left idle for long periods, however, permitting encroachment of woody vegetation and excessive build-up of litter. Further, woody species are frequently planted in grasslands.

Managers do not always state their grazing objectives, but objectives may include returning the vegetation to an assumed pristine state, favoring certain plant species or communities, reducing the accumulation of organic matter, and encouraging vegetation that supports desired wildlife species. Other objectives are economic returns and good relations with neighboring landowners who want to graze the lands. Grazing often is controversial (e.g., Fleischner 1994) because objectives often are not clearly defined and progress toward objectives may not be measured with a rigorous monitoring program (Kirby *et al.* 1992). In addition, there are numerous grazing regimes whose impacts on wildlife and habitats vary according to the type of grassland, soils, kinds of grazing animals, precipitation patterns, and other influences. It is sometimes argued that grazing by cattle represents an ecological replacement of bison grazing, but diets of the two species differ considerably (Peden *et al.* 1974, Schwartz and Ellis 1981) as do their mobility and ability to capitalize on vegetation growth over wide areas (McNaughton 1993). Moreover, the replacement of bison by cattle, and the associated fencing, reduces the heterogeneity of grazing effects and resulting habitats for some birds (Knopf 1996a).

Although grassland birds evolved with grazing animals (Knopf 1996b), the effects of grazing on birds are variable and depends on the region (Kantrud and Kologiski 1983), grazing regime, precipitation and other environmental conditions, and the species. Further, short-term effects may differ markedly from longer-term ones. Heavy grazing favors species—including burrowing owl, horned lark, and chestnut-collared longspur (Kantrud 1981)—that use shorter vegetation, but greatly reduces numbers of Sprague's pipit, sedge wren, bobolink, savannah sparrow, Baird's sparrow, Le Conte's sparrow, and common yellowthroat (Maher 1973; Owens and Myres 1973; Kantrud and Kologiski 1982, 1983; Dale 1984; Lingle and Bedell 1989). Prescott and Collister (1993) suggested that heavy grazing may reduce suitability of habitat for loggerhead shrikes in southeastern Alberta. Light grazing enhances the habitat for Baird's sparrow and clay-colored sparrow; light to moderate grazing supports higher densities of Sprague's pipit, savannah sparrow, and vesper sparrow (Kantrud and Kologiski 1983). Berkey *et al.*

(1993) suggested that short-term grazing in North Dakota likely was beneficial also to the ferruginous hawk. They deemed grazing detrimental to American bittern, northern harrier, upland sandpiper, short-eared owl, dickcissel, grasshopper sparrow, Le Conte's sparrow, and bobolink, species that prefer taller and denser grassy vegetation.

Under certain grazing systems, livestock are rotated through a series of pastures, which receive intense grazing pressure for short periods. This practice provides a patchwork of grasslands at any one time ranging from heavily grazed to idled for an entire growing season. Managers recommend these systems over season-long grazing both to enhance livestock production and to offer a variety of habitats for birds. Berkey *et al.* (1993) concluded that, as an alternative to season-long grazing, short-term rotational grazing would benefit species that favor taller and more robust vegetation, such as northern harrier, Baird's sparrow, lark bunting, and others.

Prescribed burning has similar biological, but fewer economic, objectives as grazing. Burning is made more difficult by unsafe or unsuitable weather conditions (e.g., high winds, rains), air quality concerns, personnel and equipment needs, and unfavorable attitudes by neighbors.

Burning has an immediate effect on grassland birds, although Kruse and Piehl (1986) found that 69 percent of active ground nests survived fires in mid-June. Recently burned areas are favored foraging areas for a number of species, including marbled godwits and willets. As growth resumes following a burn, habitat succession favors a sequence of species, beginning with species such as horned lark, chestnut-collared longspur, and vesper sparrow (Maher 1973, Huber and Steuter 1984, Pylypec 1991). Until the vegetation on a burned area is fully restored, the habitat is less suitable for species such as savannah sparrow, clay-colored sparrow, grasshopper sparrow, and bobolink (Tester and Marshall 1961, Halvorsen and Anderson 1980, Pylypec 1991, Herkert 1994b). Fire was a natural phenomenon in the northern plains and maintained the prairie. It may be an essential tool for managers who want to continue to maintain prairie and support true grassland birds. Fire and grazing occurred together naturally, and studies are needed of the interactive effects of these practices.

Often trees and other woody species are planted in grassland areas. Reasons for such plantings include aesthetic considerations, creation of habitat for game species such as deer and pheasants, and increased local species diversity (Cable *et al.* 1992). Even without an active planting program, encroachment by woody vegetation in grassland is favored by leaving the land idle, especially by protecting it from fire (Knopf 1994).

Grasslands invaded by woody species typically contain more bird species than those without (Arnold and Higgins 1986). These species tend to be edge or generalist species, such as brown thrasher, gray catbird, song sparrow, American robin, and common grackle. Such species have plentiful habitat elsewhere, and their populations are robust. Meanwhile, the addition of trees may reduce the quality of habitat for true grassland species, such as Sprague's pipit, Baird's sparrow, and short-eared owl. These species have much more restricted habitats or breeding ranges and require maintenance of prairie for their viability.

Woody vegetation can influence grassland birds in several ways. First, it reduces the area of grassland and fragments it; there is evidence that certain grassland birds, like some forest species, are area-sensitive. Second, it precludes certain species from using an area (Whitmore 1981, Kahl *et al.* 1985). Third, trees and shrubs provide perches for raptors and cowbirds, and travel lanes for mammalian predators. And fourth, species attracted to the woody vegetation may forage in adjacent grasslands and compete with prairie species.

Attempts are sometimes made to restore prairie after it had been cultivated (e.g., Thompson 1992). Although many native grasses and some forbs can be seeded with relative ease, prairie restoration is practical only for relatively small tracts of land. Use of restored grasslands by breeding birds is little-studied, but Blankespoor (1980) found that restored prairies in South Dakota supported breeding populations of grasshopper sparrows, dickcissels, common yellowthroats, and other species 2 to 4 years after planting. Grasshopper sparrows, Henslow's sparrows, bobolinks,

and eastern meadowlarks colonized prairie restorations in southern Wisconsin and established sizable and apparently stable populations within 4 to 5 years (Volkert 1992).

Former croplands are often replanted with mixtures of native and introduced grasses and forbs, especially legumes. These planting are made for several purposes, including providing habitat for upland-nesting ducks, enhancing soil quality, and reducing soil erosion. For example, the Soil Bank Program of the 1960's and 1970's and the Conservation Reserve Program of the 1980's and 1990's resulted in the retirement of vast areas of cultivated land and their conversion to mixtures of grasses and legumes. Such programs do not recreate natural habitats, but they do provide productive habitats for a variety of grassland birds (e.g., Duebbert 1981, Higgins *et al.* 1984, Renken and Dinsmore 1987). Johnson and Schwartz (1993) found that many species, including lark bunting, grasshopper sparrow, western meadowlark, clay-colored sparrow, bobolink, and sedge wren were relatively common in CRP fields in the northern plains, whereas they occurred far less commonly in croplands of the type that CRP replaced.

The most important practice for wetlands is protection from drainage. This goal can be achieved by purchasing in fee title, by buying easements that prevent drainage, by legal proscription, by tax incentives, or by encouraging wetland owners in other ways (Johnson *et al.* 1994a). Wetland preservation obviously protects a number of wetland-dependent species. In addition, programs such as the U.S. Fish and Wildlife Service's wetland acquisition program also acquire uplands surrounding the wetlands, which protects and restores grasslands.

Many wetlands are restored, usually by undoing the ditch or tiling used to drain the wetland (Galatowitsch and van der Valk 1994). Wetland restoration is more readily achievable than prairie restoration, but the degree to which a restored wetland performs the ecological functions of a natural wetland varies from one situation to another. Wetlands can be created even where they did not occur previously. This practice is sometimes done to mitigate for wetland losses elsewhere. Restored or created wetlands should support the customary wetland avifauna if they develop the vegetative and invertebrate communities of natural wetlands. Delphey and Dinsmore (1993) suggested that the absence of low-prairie and wet-meadow zones in restored wetlands in Iowa may have contributed to reduced bird use compared with natural wetlands. Although considerable effort has been expended in restoring wetlands in the northern plains, relatively little attention has been paid to evaluating the restorations. An extensive interagency effort is underway to remedy that situation (N. H. Euliss, Northern Prairie Science Center, pers. comm.).

Wetland managers often sought to increase the permanency or depth of wetlands, with the thought of minimizing the effects of dry seasons and years. The unfortunate consequences of this practice include reductions in desirable emergent plant communities (Kadlec and Smith 1992) and invertebrate populations. Maintaining the natural dynamics of wetlands is key to maintaining their productivity and value as habitat for birds (Weller 1978).

Some wetlands have become choked with emergent vegetation, notably hybrid cattail (*Typha x glauca*). This phenomenon typically occurs in semipermanent wetlands surrounded by cropland, where grazing and other disturbances have been eliminated (Kantrud 1992). Although dense emergent stands afford nesting habitat for species such as common yellowthroat and marsh wren, they render the wetland less suitable for waterfowl and some other species. For that reason, managers try to reduce cattail stands to create a "hemimarsh" situation (Weller and Spatcher 1965), considered the ideal condition for dabbling ducks (Kaminski and Prince 1981). Several methods of cattail reduction have been attempted, but currently the favorite is the application of the herbicide glyphosate in a patchwork pattern (Linz *et al.* 1992). Natural forces sometimes still work; heavy precipitation in 1993 and 1994 flooded out cattails throughout much of North and South Dakota.

Where water levels can be controlled in wetlands, managers attempt to achieve several goals. One is to attract birds by applying water before they arrive in spring, but drawing it down later in summer or fall. This procedure simulates the natural dynamics of seasonal wetlands, and increases productivity of plants and invertebrates. Moist soil management (e.g., Reid *et al.* 1989) is a well-studied management practice designed to mimic

natural dynamics. Originally intended to benefit waterfowl, it enhances the value of habitat for breeding and migrating shorebirds and other wetland-dependent species as well (Fredrickson and Reid 1986).

CONSERVATION STRATEGIES

Wildlife conservation has advanced from managing habitats for maximum production of game species to a more encompassing—albeit less clearly defined—objective of protecting biodiversity. Meeting that goal will require action on many levels and scales, including local, regional, state, national, and international. A focus on neotropical migrants is appropriate, in the sense that those species face special risks and have received inadequate attention in the past. Other species are of concern as well and should not be neglected simply because they winter north of the U.S.-Mexico border. Ecologists have long argued about the artificiality of political borders to wildlife; programs such as Partners in Flight should be used to minimize barriers, not reinforce them.

We also must be clear about the meaning of biodiversity. Preserving diversity means protecting the various forms of life and the habitats and processes that support them (Keystone Center 1991). It does not mean maximizing local species diversity. Adding trees to a prairie landscape, for example, will increase local biodiversity by providing new habitat for such species as brown thrasher, gray catbird, song sparrow, common grackle, and western kingbird. But it will not enhance their viability, for these species are widespread, common, and can thrive independently of happenings on the prairie. Conversely, the addition of trees to grasslands may reduce the viability of true prairie birds, species whose future does depend on the grasslands (e.g., Knopf 1994).

Further, we should contrast management of prairie landscapes from that of cropland landscapes. Highly cultivated areas are generally depauperate of bird species, except certain "weedy" ones or those that favor sparse cover (Best *et al.* 1995). In such situations, adding trees and shrubs, and managing roadsides for wildlife will enhance the local species diversity and provide a more aesthetically pleasing environment without negatively affecting other species. In a true prairie situation, however, those same practices could be detrimental.

One promising development is the pairing of two major conservation partnerships, the North American Waterfowl Management Plan, whose goal is the restoration of waterfowl populations and habitats in North America, and the Partners In Flight program, which emphasizes neotropical migrant landbirds. The Waterfowl Management Plan works primarily through geographically based joint ventures. The Lower Mississippi Valley Joint Venture recently completed a habitat plan that encompasses neotropical migrant birds and shorebirds, as well as waterfowl. The Prairie Pothole Joint Venture has begun a similar effort. The initial emphasis will be to identify species of special concern, determine how management activities affect those species, and propose appropriate management strategies.

Unlike the situation with the Waterfowl Management Plan, defining population objectives for nongame species will be virtually impossible. Reasonable estimates of population size for those species are almost totally lacking. The BBS monitors trends in certain populations with some accuracy, but is imperfect for many others. Further, the often-low philopatry (McNicholl 1988) in many grassland and wetland birds argues against area-specific objectives. For example, Conservation Reserve Program fields in Eddy County, North Dakota, supported an average of only 0.03 pairs of Le Conte's Sparrows per 100 ha during 1991-1993. In 1994, because of extremely wet conditions, the density jumped to 21.47. Those conditions persisted into 1995, when the density continued to climb to 73.27 (unpubl. data). Other grassland and wetland species exhibit similar dynamic responses to precipitation, wetland conditions, temperature, and local land use such as burning and grazing. Establishing specific population objectives for particular habitats is not feasible; our approach should be to provide the habitat base that—when other environmental conditions are right—will support desired and sustainable populations.

SOME MANAGEMENT RECOMMENDATIONS

The following thoughts are offered regarding what could be done in the northern prairies to

enhance habitat for migratory birds. The proposed actions would be beneficial to a variety of other species as well, and would offer general conservation benefits.

Save the Sod.—Perhaps the highest priority is maintaining the base of native grassland that still remains. Several species of birds absolutely require this habitat. Most native grassland is privately owned, and much of it is excessively grazed. Nonetheless, even overgrazed prairie provides better habitat for grassland birds that does the alternative, cultivated fields. Although it would be worthwhile for agencies and conservation organizations to purchase native grasslands and manage them for their natural values, the total area that could be protected in this manner pales in comparison to the needs. Thus, migratory birds—and those who care about migratory birds—are dependent on privately owned ranches and farms. Maintaining range- and pasture lands is a conservation priority, and individuals and groups whose interests include birds should cooperate with ranching and farming advocates on issues of mutual benefit. Too often, the groups view one another as antagonists, rather than potential collaborators.

Grassland can be restored, but the full complement of forbs, vertebrate animals, invertebrates, and soil microorganisms cannot. It is far less costly to maintain a prairie than to reconstruct one. Further, existing grasslands often can be managed differently, in ways that improve the habitat they afford wildlife while still providing economic returns to the landowner. Biologists, range scientists, and the agricultural community should work together to develop and test land-use practices that offer economic benefits as well as wildlife and other conservation values. These practices might include rest-rotational grazing systems, minimum-till agriculture, and integrated pest management, among others.

Preserving wetlands will also benefit many species of migratory birds. In contrast to prairie, restoration of wetlands can be accomplished easier and with greater success. Wetlands have positive non-wildlife values, such as water retention to reduce downstream flooding, but also negative values, as impediments to agricultural operations, for example.

Target Farm Programs for Conservation Benefits.—Billions of dollars have been expended under past farm programs to balance supplies of commodities with demands and to maintain farm economies. Some of those programs also afforded conservation benefits, including protection from soil erosion and habitat for wildlife, but many did not. Although the current "Freedom to farm" plan appears to foretell the end of farm programs, it remains to be seen if that result will be realized. In any event, future farm programs could be developed to include conservation and wildlife benefits as high priorities. As one example, long-term rather than short-term set-asides not only permit cover to be planted that will benefit wildlife, they also help farmers plan with greater certainty about the future.

Manage What We Have.—Public lands, including national wildlife refuges, waterfowl production areas, national and state parks, national grasslands, and game management areas, are managed in various ways. Too often, little is known about the effects of those management practices on wildlife. Researchers and managers need to work together to learn about those effects. Researchers should not avoid management questions because they are "too applied," and managers should not avoid evaluating practices they apply because the research "costs too much and takes too long." The results of moving dirt are immediate; results of an evaluation are longer in coming, but may have more lasting value.

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MANAGEMENT OF AGRICULTURAL LANDSCAPES FOR THE CONSERVATION OF NEOTROPICAL MIGRATORY BIRDS

Rolf R. Koford[1] and Louis B. Best[2]

ABSTRACT.—Thirty-eight neotropical migratory birds (NTMBs) are common in the agricultural landscapes of the Midwest. Most of these species depend on herbaceous or wooded habitats, which are declining as the average size of farms in this region increases about 12 percent per decade. We recommend minimizing cultivation; encouraging moderate grazing; delaying spring mowing of hayfields, grassed waterways, and roadsides; avoiding nighttime mowing; encouraging longer intervals between mowings; avoiding fall mowing and annual mowing of grassed waterways and roadsides; retaining fencelines along roadsides; maintaining idle land that provides nesting cover; and changing the amount and configuration of idle land to enhance species richness and nesting success.

INTRODUCTION

Agriculture is the dominant land use in the Midwest. Farmland composed 66 percent of the land area in 12 midwestern states (listed in table 2) in 1993 (USDA 1993). In 1992, half of the rural, non-federal land in the Midwest was active cropland, a quarter was pasture or rangeland, 17 percent was forest or woodland, and the rest was farmsteads, shelterbelts, idle cropland, etc. (USDA 1994). Rodenhouse *et al.* (1993, 1995) reviewed the effects of agriculture on NTMBs. Here we focus on these effects in the Midwest.

NTMBs constitute the major part of midwestern, farmland avifaunas. Thirty-eight NTMB species are common in at least one farmland habitat in the Midwest, and two-thirds of the common native species are NTMBs (table 1). Thus the conversion of midwestern forest, savannah, and prairie to agriculture, mostly prior to 1920, undoubtedly changed the abundance and distribution of NTMBs. Changes in bird abundance are not well documented because there were no widespread, systematic surveys of bird populations before the mid-1960's. Extinction and local extirpation of some species have been documented (e.g., Dinsmore 1981, 1994). Grassland birds, which form a large fraction of the common farmland NTMBs (table 1), evidently did not decline much between the 1920's and 1950's, when diversified farming was common (Warner 1994). The modern era of intensive cropping coincides with population declines of grassland birds (Herkert 1991, 1995; Warner 1994). Other long-term changes, mostly of declining bird populations, have been noted in Illinois (Graber and Graber 1963) and a small study area in Iowa (Lowther 1984). These declines have continued in recent years (Graber and Graber 1983, Warner 1994). A variety of factors, acting at all stages of the annual cycle, may be contributing to these population declines.

Considering the extent of modern agriculture, the effect that agriculture has had on bird populations, and the likelihood of continuing changes, conservation biologists need to understand how future changes in agriculture are likely to affect bird populations. Will some species continue to decline even without further changes in the landscape? Which species are most vulnerable? To address these questions and others, we review how agriculture has changed recently, how agricultural land-use practices affect particular bird populations, and what can be done to conserve species likely to be negatively affected by current and future practices.

[1] *Iowa Cooperative Fish and Wildlife Research Unit, Iowa State University, Ames, IA 50011-3221*

[2] *Department of Animal Ecology, Iowa State University, Ames, IA 50011-3221*

Table 1.—*Migratory status and habitat affinities of bird species that are (1) common in at least one habitat in agricultural landscapes; (2) midwestern species of management concern due to rarity, habitat specialization, or population trend; or (3) both. Habitat affinity codes are C (common) and x (use). (Cropland includes small grains, row crops, and hayland.)*

Common species[1] and species of concern[2]	Migratory status[3]	Crop-land	Herbaceous habitats	Wooded habitats	Wet-land
Common loon (*Gavia immer*)[2]	W				x
American bittern (*Botaurus lentiginosus*)[2]	W				x
Least bittern (*Ixobrychus exilis*)[2]	W				x
Trumpeter swan (*Cygnus buccinator*)[2]	W				x
Mallard (*Anas platyrhynchos*)	W				C
Blue-winged teal (*Anas discors*)	W				C
Northern shoveler (*Anas clypeata*)	W				C
Gadwall (*Anas strepera*)	W				C
Northern harrier (*Circus cyaneus*)[2]	B		x		
Northern goshawk (*Accipiter gentilis*)[2]	B			x	
Red-shouldered hawk (*Buteo lineatus*)[2]	B			x	
Peregrine falcon (*Falco peregrinus*)[2]	A		x		
Gray partridge (*Perdix perdix*)[4]	R		C		
Ring-necked pheasant (*Phasianus colchicus*)[4]	R		C	C	
Northern bobwhite (*Colinus virginianus*)	R			C	
Yellow rail (*Coturnicops noveboracensis*)[2]	W				x
Black rail (*Laterallus jamaicensis*)[2]	W				x
Virginia rail (*Rallus limicola*)	W				C
American coot (*Fulica americana*)	W				C
Upland sandpiper (*Bartramia longicauda*)[2]	A		x		
Common tern (*Sterna hirundo*)[2]	W				x
Least tern (*Sterna antillarum*)[2]	W				x
Black tern (*Chlidonias niger*)[2,5]	W				C
Rock dove (*Columba livia*)[4]	R			C	
Mourning dove (*Zenaida macroura*)	B			C	
Black-billed cuckoo (*Coccyzus erythropthalmus*)	A			C	
Yellow-billed cuckoo(*Coccyzus americanus*)[2]	A			C	
Barn owl (*Tyto alba*)[2]	R			x	
Burrowing owl (*Speotyto cunicularia*)[2]	A		x		
Short-eared owl (*Asio flammeus*)[2]	B		x		
Red-headed woodpecker (*Melanerpes erythrocephalus*)[2]	S			C	
Downy woodpecker (*Picoides pubescens*)	R			C	
Northern flicker (*Colaptes auratus*)[2]	B			C	
Olive-sided flycatcher (*Contopus borealis*)[2]	A			x	
Eastern wood-pewee (*Contopus virens*)	A			C	
Eastern kingbird (*Tyrannus tyrannus*)	A		C	C	
Horned lark (*Eremophila alpestris*)	B		C		
Barn swallow (*Hirundo rustica*)	A			C	
Blue jay (*Cyanocitta cristata*)	S			C	
American crow (*Corvus brachyrhynchos*)	S			C	
Black-capped chickadee (*Parus atricapillus*)	R			C	
White-breasted nuthatch (*Sitta carolinensis*)	R			C	
Bewick's wren (*Thryomanes bewickii*)[2]	S			x	
House wren (*Troglodytes aedon*)	A			C	
Sedge wren (*Cistothorus platensis*)[2]	B		C		C
Marsh wren (*Cistothorus palustris*)	B				C
Veery (*Catharus fuscescens*)[2]	A			x	

(*table 1 continued on next page*)

(*table 1 continued*)

Common species[1] and species of concern[2]	Migratory status[3]	Crop-land	Herbaceous habitats	Wooded habitats	Wet-land
Wood thrush (*Hylocichla mustelina*)[2]	A			x	
American robin (*Turdus migratorius*)	B		C	C	
Gray catbird (*Dumetella carolinensis*)	A			C	
Brown thrasher (*Toxostoma rufum*)	S			C	
Loggerhead shrike (*Lanius ludovicianus*)[2]	B			C	
European starling (*Sturnus vulgaris*)[4]	R			C	
Bell's vireo (*Vireo bellii*)	A			x	
Blue-winged warbler (*Vermivora pinus*)	A			x	
Golden-winged warbler (*Vermivora chrysoptera*)	A			x	
Yellow warbler (*Dendroica petechia*)[5]	A			C	
Blue-winged warbler (*Vermivora pinus*)	A			x	
Golden-winged warbler (*Vermivora chrysoptera*)	A			x	
Common yellowthroat (*Geothlypis trichas*)	A		C	C	C
Northern cardinal (*Cardinalis cardinalis*)	R			C	
Rose-breasted grosbeak (*Pheucticus ludovicianus*)	A			C	
Indigo bunting (*Passerina cyanea*)	A			C	
Dickcissel (*Spiza americana*)[2]	A	C	C	C	
Rufous-sided towhee (*Pipilo erythrophthalmus*)[5]	B			C	
Bachman's sparrow (*Aimophila aestivalis*)[2]	S			x	
Chipping sparrow (*Spizella passerina*)	A			C	
Clay-colored sparrow (*Spizella pallida*)[5]	A		C		
Field sparrow (*Spizella pusilla*)[2]	S			C	
Vesper sparrow (*Pooecetes gramineus*)	B		C	C	
Lark bunting (*Calamospiza melanocorys*)[5]	A		C		
Savannah sparrow (*Passerculus sandwichensis*)	B		C		
Baird's sparrow (*Ammodramus bairdii*)[2,5]	A		C		
Grasshopper sparrow (*Ammodramus savannarum*)[2]	A	C	C	C	
Henslow's sparrow (*Ammodramus henslowii*)[2]	S		x		
Song sparrow (*Melospiza melodia*)	B		C	C	
Swamp sparrow (*Melospiza georgiana*)	B				C
Chestnut-collared longspur (*Calcarius ornatus*)[5]	B		C		
Bobolink (*Dolichonyx oryzivorus*)[2]	A	C	C		
Red-winged blackbird (*Agelaius phoeniceus*)	B	C	C	C	C
Eastern meadowlark (*Sturnella magna*)[2]	B		C	C	
Western meadowlark (*Sturnella neglecta*)	B		C	C	
Yellow-headed blackbird (*Xanthocephalus xanthocephalus*)	A				C
Common grackle (*Quiscalus quiscula*)	S		C	C	C
Brown-headed cowbird (*Molothrus ater*)	B		C	C	
Northern oriole (*Icterus galbula*)	A			C	
American goldfinch (*Carduelis tristis*)	B			C	
House sparrow (*Passer domesticus*)[4]	R		C	C	

[1] The list of common species is from Best et al. (1995) unless otherwise indicated. Common species were those in which densities have exceeded 25 birds per 100 ha.

[2] USFWS (1995).

[3] A: breed in U.S. or Canada, winter south of U.S.
B: breed in U.S. or Canada, winter in U.S. and south of U.S.
S: short-distance migrants; not neotropical migrants (Gauthreaux 1992).
R: year-round resident species.
W: wetland bird, therefore not on the preliminary list of neotropical migratory landbirds (Gauthreaux 1992).

[4] Exotic species.

[5] Common according to Stewart and Kantrud (1972).

RECENT CHANGES IN AGRICULTURAL HABITATS AND LANDSCAPE STRUCTURE

The changes in agricultural landscapes that most concern conservation biologists are those that have occurred on a large spatial scale. We focus on changes in agricultural habitats in the last three decades, when bird population trends in the Midwest were well documented (Herkert 1995). The major habitats in the Midwest are crops, including small grains (wheat, oats, etc.), corn, soybeans, sunflowers, and forage (hay). Other habitats are pastureland and rangeland, wetlands, woodlands, and strip cover (e.g., fencerows, roadsides, waterways, terraces). Idle cropland is not explicitly considered a habitat in this section but is covered below under land-use practices. The amounts and relative proportions of cropland, pastureland and rangeland, woodland, and other rural land vary across the Midwest (table 2).

The total area of cropland in the Midwest in the early 1980's was similar to that in the late 1940's (Frey and Hexem 1985). There was a steady decline in cropland area, associated with surplus production, from the late 1940's to the mid-1970's. The area then increased until the early 1980's, as exports expanded. The pattern of change varied geographically. The area used for crops in the Northern Plains (Kansas, Nebraska, North Dakota, South Dakota) in the early 1980's was similar to that in the late 1940's. In the Corn Belt (Illinois, Indiana, Iowa, Missouri, Ohio), there was a dip in the mid-1970's but an increase of 9.5 percent between the late 1940's and early 1980's. In the Lake States (Michigan, Minnesota, Wisconsin), there was an increase of 2.1 percent over the same period. The area of active cropland in the Midwest declined again from 1982 to 1992, mostly due to enrollment of cropland in the Conservation Reserve Program (CRP; USDA 1994).

Analysis of data from the National Resources Inventory (NRI), which has been conducted since 1967, also demonstrates declines in cropland area in the Midwest between 1967 and 1977 and between 1982 and 1992 (fig. 1). Analysis procedures changed between 1977 and 1982, so inventory data before 1977 cannot be directly compared with data after 1982 (USDA 1990).

Table 2.—*Estimated land use (thousands of hectares) of non-federal, rural land in the 12 midwestern states in 1992 (USDA 1994). Pastureland includes rangeland and excludes grazed forested land. Land enrolled in the Conservation Reserve Program is included with other rural land.*

Region State	Cropland	Pastureland	Woodland	Other
Corn Belt				
Illinois	9,761	1,119	1,385	563
Indiana	5,473	756	1,469	481
Iowa	10,120	1,503	782	1,196
Missouri	5,406	4,875	4,721	944
Ohio	4,831	919	2,683	516
Total	35,590	9,173	11,039	3,701
Lake States				
Michigan	3,639	953	6,321	975
Minnesota	8,649	1,329	5,595	2,520
Wisconsin	4,379	1,196	5,431	1,301
Total	16,667	3,479	17,347	4,796
Northern Plains				
Kansas	10,759	7,302	539	1,444
Nebraska	7,792	10,017	315	865
North Dakota	10,021	4,655	173	1,693
South Dakota	6,657	9,757	219	1,319
Total	35,228	31,731	1,245	5,321

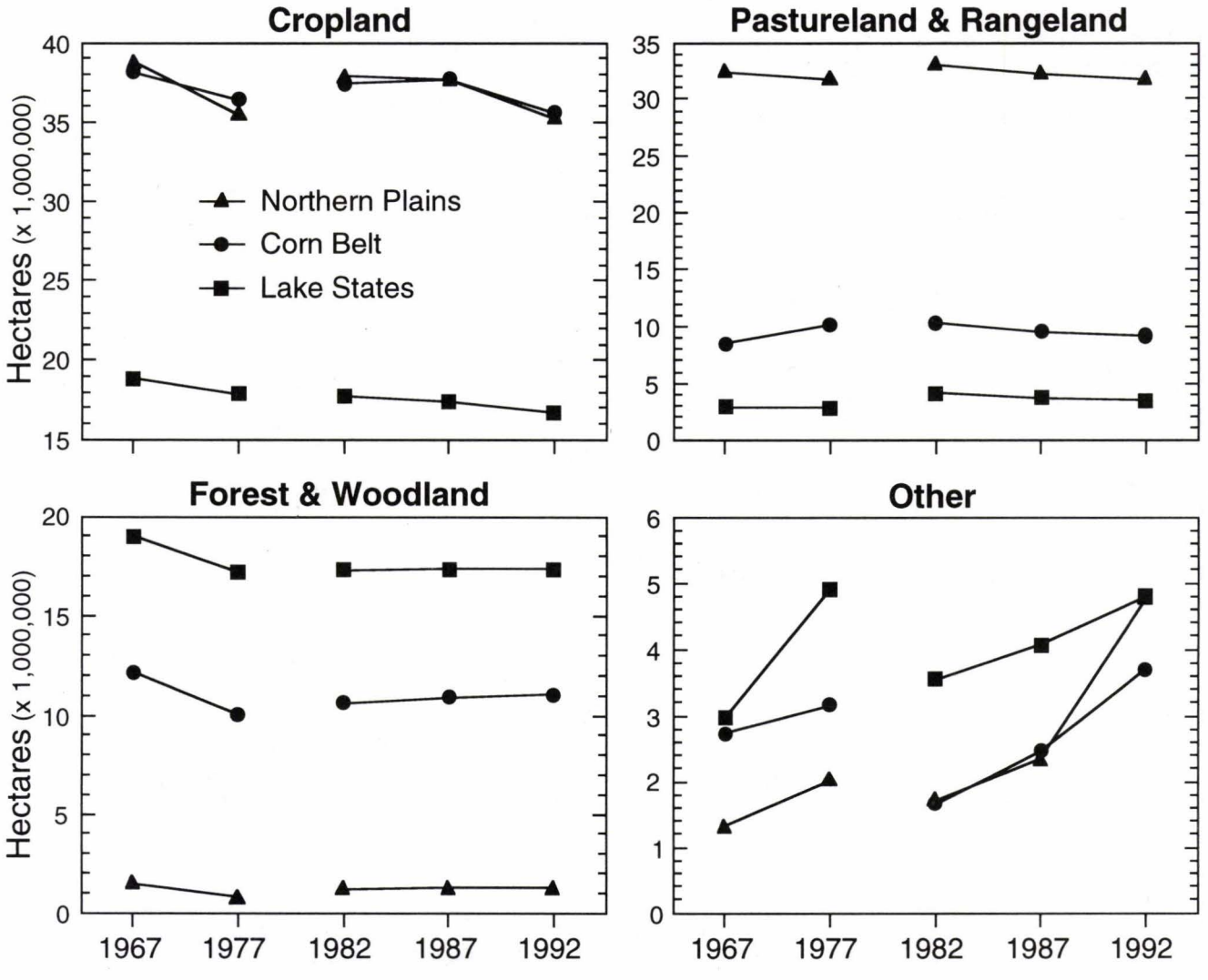

Figure 1.—*Recent changes in estimated land use of non-federal, rural land in three midwestern regions (USDA 1971b, 1982a, 1994). "Pastureland & Rangeland" excludes grazed forested land. Analysis procedures changed between 1977 and 1982, so inventory data before 1977 cannot be directly compared with data after 1982 (USDA 1990).*

With the exception of the CRP, changes in the total area of midwestern cropland have been relatively small, particularly during the last three decades, when the best bird survey data have been available. We focus on these decades in our presentation of changes at the level of major habitats.

Cropland Habitats

Small grains.—Wheat and other small grains are grown mostly in the Northern Plains. Wheat area increased in the 1970's and has remained relatively stable since then (fig. 2). The area planted to oats has declined by two thirds over the past three decades.

Corn and soybeans.—The area planted to corn in the Midwest declined in the 1960's, increased in the 1970's, then declined in the late 1980's (fig. 2). Soybean area increased steadily from the 1960's until the 1980's, then remained fairly stable. Corn and soybean area in Illinois reflected this pattern and increased from 4.9 million ha in 1945 to 8.4 million ha in the 1980's (Warner 1992). Corn and soybeans are usually grown in alternate-year rotation.

Sunflowers.—Sunflowers are grown mostly in North Dakota, South Dakota, and Minnesota (USDA 1993). Sunflowers are a relatively new crop in these states (fig. 2). The rapid increase

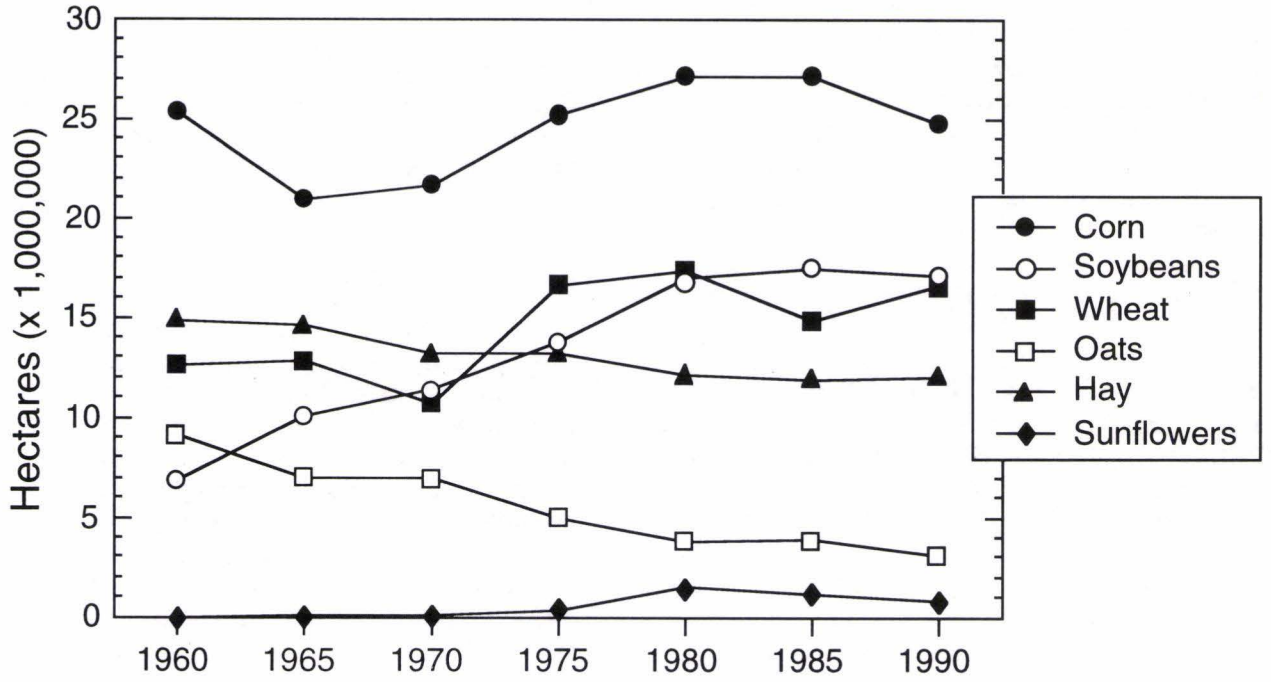

Figure 2.—*Recent changes in planted area of selected crops and harvested area of hay in the Midwest (USDA 1962, 1967, 1972, 1975, 1977, 1982b, 1987, 1991).*

in sunflower area was followed by a decline associated with increased worldwide production (McCormick *et al.* 1992).

Hayland.—After a decline in the late 1960's, the area of harvested hay in the Midwest has remained stable (fig. 2). Traditionally, livestock provided agricultural diversity for farmers who also raised crops, and hay was harvested from untilled prairie which was sometimes seeded with grasses. On tilled land, alfalfa is the primary hay crop. Herkert *et al.* (1996) discuss hayland further.

Non-cropland Habitats

Pastureland and Rangeland.—The general trend in the Midwest between 1982 and 1992 was toward less pastureland and rangeland; between 1967 and 1977 the trend varied geographically (fig. 1). Herkert (1995) reported a 53 percent decline in pastureland from 1964 to 1992 in Illinois, Iowa, Ohio, and Wisconsin. Pastureland is used primarily for production of forage plants for livestock and is typically managed by fertilization and reseeding, according to the instructions for reporting NRI data. Rangeland is land on which the climax or potential vegetation is grasses, forbs, and shrubs suitable for grazing; management does not usually include application of chemicals. Pastureland is more common in the eastern part of the Midwest and rangeland is more common in the western part. In Iowa, the amount of pastureland declined by 23 percent between 1958 and 1967 (Iowa Soil and Water Conservation Needs Inventory Committee 1970). Most of this converted Iowa land was classified as forest or cropland in 1967. At a national scale, too, much of the conversion to cropland prior to the 1980's was from rangeland and pastureland (Heimlich 1985).

Wetlands.—Wetlands are a prominent part of the natural landscape in the Prairie Pothole Region of North Dakota, South Dakota, Minnesota, and northern Iowa. Many wetlands have been connected to ditches or underground pipes (tile lines) to facilitate rapid drainage of cropland in the spring. Loss of original wetland area in the Midwest ranged from 164,211 ha in Kansas to 2,816,802 ha in Illinois (Dahl 1990). The percentage of wetland area lost ranges from 35-48 percent in the Northern Plains states, 42-50 percent in the Great Lakes states, and 85-90 percent in the Corn Belt. Wetlands are discussed further by Johnson (1996).

Woodlands.—Woodlands on midwestern farms are mostly riparian areas, woodlots, and shelterbelts. They are more abundant in the east (table 2). In Iowa, the amount of land classified as forest or woodland increased between 1958 and 1967 (Iowa Soil and Water Conservation Needs Inventory Committee 1970). Most of this land had previously been classified as pasture. Woodland habitat decreased in the Midwest from 1967 to 1977 but has been relatively stable since then (fig. 1). Below, we discuss shelterbelts as a form of strip cover under land-use practices. We do not consider riparian areas and woodlots further in this paper. Wooded, non-strip-cover habitats may be the only nesting habitats used by some woodland species, such as cerulean warblers (see other chapters, this volume).

Strip cover.—Some strip-cover habitats (e.g., fencerows) have declined in recent decades as the average size of crop fields has increased. Rodenhouse *et al.* (1993) reviewed information on loss of fencerows in midwestern agricultural regions and found that 30-80 percent of fencerows had been removed since the 1930's.

Landscape structure.—The habitats discussed here (small grains, corn, soybeans, sunflowers, hay, pastureland and rangeland, wetlands, woodlands, and strip cover) are arranged in landscapes that affect NTMBs (Freemark *et al.* 1995). Interspersion of habitat types, habitat-patch size, and other aspects of landscape structure can affect species composition, abundance, pairing and reproductive success, and population dynamics. Species richness and abundance of birds in farmland are greatest in grasslands, pasture, early-successional habitats, strip cover, and shelterbelts (Freemark *et al.* 1995). Recent declines in the areal extent of these habitats have led to a simplification, or reduction in diversity, of farmland landscapes in general (Warner 1994). In Ohio, for example, the percentage of farmland that was harvested increased from 45 to 67 percent between 1940 and 1982 (Barrett *et al.* 1990). The declines in these habitats also have been accompanied by a trend toward larger crop fields. As an indication of this trend, the average size of farms in the Midwest in 1970, 1980, and 1990 was 164, 183, and 206 ha, respectively (USDA 1971a, 1981, 1991).

EFFECTS OF LAND-USE PRACTICES ON NTMBs

The agricultural habitats that appear to have declined the most are pastureland and strip cover. These declines may have contributed to population declines of grassland and shrubland birds (Herkert 1995). Population declines also may be related to agricultural land-use practices. Land-use practices create, modify, and destroy habitat for various species. These practices include cultivation, chemical application, grazing, haying, idling cropland, and maintaining strip cover. Some of these practices have changed in recent years, but it is important to realize that a practice may be contributing to population declines (e.g., by providing poor nesting habitat) even if the extent of the practice has not changed. Here we consider each practice individually. The effects of such practices in combination (i.e., farmland structure) is covered elsewhere (Freemark *et al.* 1995, Rodenhouse *et al.* 1995).

Cultivation

Cultivation, or tillage, of soil is integral to the process of raising most crops. The amount of cultivation varies with the tillage system. Conventional tillage consists of tilling fields up to several times per year to prepare fields for planting and to control weeds after the crop has emerged. Little crop plant residue remains on the surface of the soil during the growing season. Tillage is often the main method of weed control in organic farming. Conservation tillage, defined by the Conservation Technology Information Center (CTIC), is any tillage and planting system that maintains at least 30 percent of the soil surface covered by residue after planting to reduce soil erosion by water or, where wind erosion is the primary concern, maintains at least 1,000 pounds/acre of residue throughout the critical wind-erosion period (CTIC 1994). The types of conservation tillage are no-till, ridge-till, and mulch-till. Reduced tillage, which leaves 15-30 percent crop residue or 500-1000 pounds/acre of residue, is a form of crop residue management but is not currently considered conservation tillage (CTIC 1994).

Cultivation affects birds in several ways (Best 1985, Rodenhouse *et al.* 1993). The amount of cultivation of rowcrop fields influences bird use by affecting the amount of crop residue on the surface of the soil (Rodenhouse and Best 1983, Basore *et al.* 1986). An immediate effect of cultivation may be to expose arthropods and other prey to foraging birds, etc. A more lasting effect, however, is a reduction in abundance of the litter-dwelling arthropods that are prey items for many birds. Conventionally tilled fields have lower arthropod abundance than no-till fields or idle areas except during pest outbreaks in the crop (Hendrix *et al.* 1986). Early summer cultivation also can disrupt nesting activity, destroying nests or causing nest abandonment (e.g., Rodenhouse and Best 1983).

The extent of conservation tillage has increased in recent years. Using a slightly different definition of conservation tillage from that in CTIC (1994), CTIC (1983) reported that conservation tillage was used on 10 percent of U.S. cropland in 1982 and reduced tillage on another 14 percent. Conservation tillage increased from 28 to 40 percent of the planted area in the Midwest between 1990 and 1994 (CTIC 1994). This increase occurred throughout the Midwest (fig. 3). Reduced tillage was used on another 24 percent of the planted area in the Midwest in 1994 (CTIC 1994). This increase in conservation tillage is an important contribution to sustainable agriculture (Barrett *et al.* 1990). Other contributions are organic farming, alternative crops, and strip intercropping.

Castrale (1985) compared conventionally tilled fields (corn and soybeans) with no-till fields in Indiana and found one third more bird species on the no-till fields during the summer. No-till fields had greater total bird abundance, and the eight most common species were more abundant on no-till fields. Killdeer and horned larks were more abundant on conventionally tilled fields. No-till fields are also more attractive than tilled fields to most NTMBs in Iowa (Basore *et al.* 1986) and Illinois (Warburton and Klimstra 1984).

Chemical Application

Application of herbicides can affect birds by reducing the availability of seeds. Herbicides and insecticides reduce the abundance and diversity of litter- and foliage-dwelling arthropods (Rodenhouse *et al.* 1993). Pesticides also can cause acute or sublethal effects on birds. Gard and Hooper (1995) reviewed

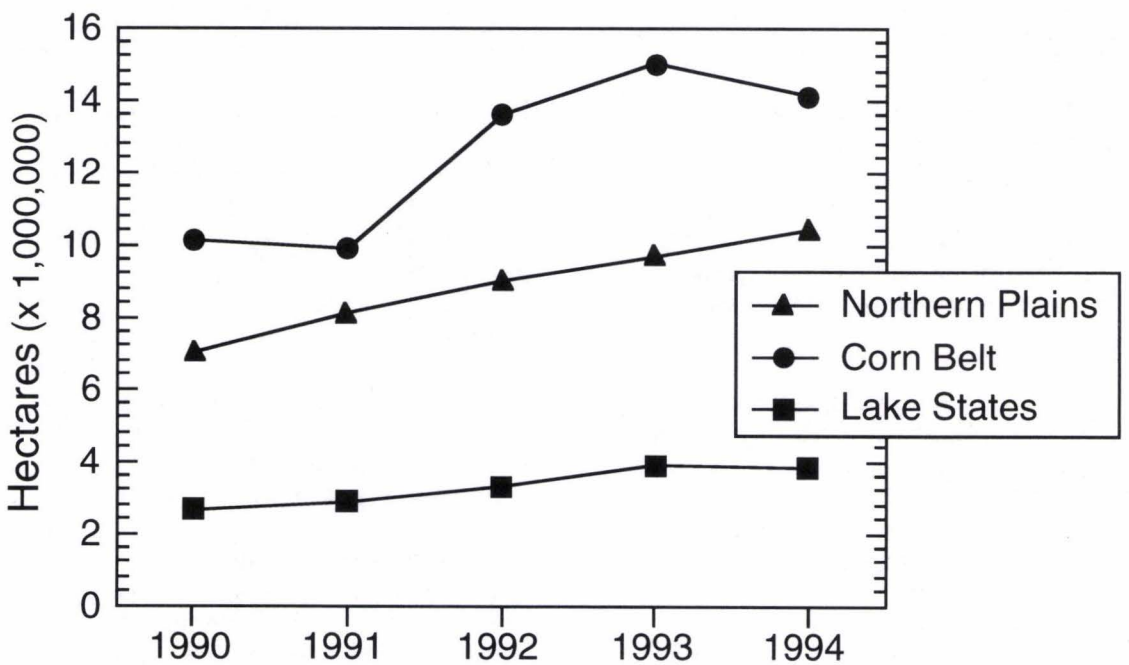

Figure 3.—*Recent changes in estimated extent of conservation tillage in three midwestern regions (CTIC 1994).*

the effect of pesticides and contaminants on NTMBs and pointed out that the effects on populations are poorly understood. Insecticides and contact herbicides are the main methods of weed and insect control in no-till fields (Castrale 1985, Wooley *et al.* 1985).

Grazing

Grazing by livestock can have direct and indirect effects on NTMBs. Nests can be lost from trampling if stocking rates are high (Jensen *et al.* 1990). Grazing also can affect the avifauna composition by altering vegetation structure (Skinner *et al.* 1984). Grazing effects are discussed further by Herkert *et al.* (1996).

Haying

Mowing hay drastically alters the structure of the vegetation, which affects species differently depending on their habitat preferences (Frawley and Best 1991). Mowing hay also can cause nest losses as well as mortality of fledglings and adults (Frawley 1989, Rodenhouse *et al.* 1993). If mowing is frequent, many birds may not be able to complete their nesting cycles. Haying is discussed further by Rodenhouse *et al.* (1993, 1995) and Herkert *et al.* (1996).

Idling Cropland

Little farmland in the Midwest is idled unintentionally because the land is generally very productive. In 1987, 11 percent of the farmland in the midwestern states was idle cropland (USDA 1993). Most of this land was idled by federal cropland-retirement programs. Annual set-asides are used to adjust the planted area in an attempt to influence the supply of various crops. Also, there have been multiple-year retirement programs, such as Soil Bank and the CRP (Harmon and Nelson 1973, Berner 1988).

Few NTMBs are likely to benefit from annual set-asides. On annual set-asides in Illinois (R. Warner, Univ. of Illinois, Urbana, pers. comm.) and Iowa (L. B. Best, pers. observ.), the most commonly planted vegetation is oats (*Avena sativa*), in which few bird species nest (Best *et al.* 1995). Furthermore, the value of this vegetation as nesting habitat is limited if it is planted late in the nesting season because nesting cover is not established when breeding begins. For example, Warner (pers. comm.) found that, in the 1980's, very little hay and oats persisted over the winter; therefore, very little of this vegetation was well established in the spring. In contrast, half or more of this vegetation persisted over the winter in the 1960's and early 1970's. An additional limitation in this habitat is that mowing destroys some of the nests that are initiated. Finally, the great spatial and temporal variability of this habitat means that philopatric birds may not find suitable nesting habitat in the same place from year to year.

Land enrolled in multiple-year retirement programs, on the other hand, generally has significant value for wildlife species (Harmon and Nelson 1973). Recent work has documented use of CRP fields by NTMBs. Surveyed CRP fields in western Minnesota, North Dakota, South Dakota, and eastern Montana had high species richness and high abundance of some NTMBs compared with cropland (Johnson and Schwartz 1993a). Many of the species that were more abundant in CRP fields had declined in the central U.S. in the quarter century before the CRP era (Johnson and Schwartz 1993b). Lark buntings and grasshopper sparrows, for example, were the most abundant species in CRP fields, and their populations declined substantially during this period. In contrast, species that were more abundant in cropland than in CRP fields had been relatively stable before the CRP era. These findings suggest that the CRP may have benefited many NTMBs that nest in grassland, assuming that the CRP boosted reproduction.

Nest success in CRP fields is comparable or higher than that in some other habitats. Bird use of CRP fields (exotic, cool-season grasses) differed from use of reduced-tillage, rowcrop fields (Patterson and Best 1996). The most abundant species in CRP fields were red-winged blackbirds, dickcissels, grasshopper sparrows, and bobolinks. Horned larks and vesper sparrows were the only species found nesting in rowcrop fields. In CRP fields, nest success (Mayfield estimate) was 34 percent for ring-necked pheasants, 14 percent for dickcissels, 16 percent for vesper sparrows, 30 percent for grasshopper sparrows, and 15 percent for red-winged blackbirds (based on at least 27 active nests for each of these five species). These estimates were higher, in general, than estimates from alfalfa fields and strip-cover habitats, indicating that CRP fields are better nesting habitat than many of the alternative habitats (Patterson and Best 1996).

Nest success in CRP fields also may be comparable to that in pastureland, although some CRP fields attract fewer species and individuals than some pastureland. Nest success did not differ significantly between CRP fields (native grasses) and grazed, native-vegetation pastures in Kansas (Granfors 1992, Klute 1994). Nest success (Mayfield estimate) was 11 to 25 percent in eastern meadowlarks and 6 to 12 percent in dickcissels. Avian abundance in native-grass CRP fields was lower than that in moderately grazed, native-vegetation pastures (Klute 1994). Other CRP fields in the same area, however, had greater grass cover and greater bird abundances than those used in the pasture comparison (Hull 1993).

It is important to recognize that the habitat structure of CRP fields changes over time. Older CRP fields in Michigan tended to have lower avian abundance and diversity (Millenbah 1993). The cover on older fields was more dense and these fields had less bare ground than younger fields.

Maintaining Strip Cover

Strip cover provides habitat for most of the NTMBs of agricultural landscapes because it is used by birds requiring either herbaceous or woody vegetation (table 1). These areas, which include grassed waterways, terraces, fencerows, roadsides, and windbreaks/shelterbelts, usually provide habitat that is more long term than that provided by areas enrolled in cropland-retirement programs. Food (e.g., arthropods, weed seeds) is often more abundant in strip cover than in crop fields, and complex vegetation structure provides nesting sites, song perches, and cover (Rodenhouse *et al.* 1993). Many of these habitats are associated with conservation practices used to control soil erosion, such as crop rotation, terraces, contour planting, strip intercropping, grassed waterways, and windbreaks. Some kinds of strip cover are subject to disturbances such as mowing. During the past few decades, the average size of cropped fields has increased, reducing the amount of strip-cover habitat (Rodenhouse *et al.* 1995).

Grassed waterways are heavily used by birds. These waterways are natural drainage systems or channels constructed to transport water off crop fields at a non-erosive velocity. Various grass species are planted in these waterways to slow the flow of water. In Iowa, 48 bird species used grassed waterways, compared with only 14 in surrounding rowcrop fields (Bryan and Best 1991). The most abundant bird species using waterways were red-winged blackbirds, dickcissels, barn swallows, grasshopper sparrows, brown-headed cowbirds, song sparrows, and western meadowlarks. Total bird abundance in waterways averaged 2,198 birds/census count/100 ha, compared with 682 for crop fields. Eleven bird species nested in the grassed waterways; red-winged blackbird and dickcissel nests were most common (Bryan and Best 1994). Considering nests of all species, more than twice as many nests were found in forbs as in grass. Sweetclover (*Melilotus* spp.) and curly dock (*Rumex crispus*) were predominant nest substrates. Orientation of crop rows relative to the waterway influenced agriculture-related disturbance. Nest densities were greater when the rows paralleled the waterways. Nest success (Mayfield estimate) of red-winged blackbirds and dickcissels in waterways was 8.4 and 22.0 percent, respectively. Predation was the most common cause of nest loss (57 percent of all nest losses), followed by mowing (16 percent).

The practice of planting and maintaining grassed waterways clearly provides habitat for a variety of NTMBs. Mowing, however, affects the quality of this habitat. Mowing alters the structure of the vegetation, which in turn may affect the bird community. Dickcissels, common yellowthroats, and red-winged blackbirds preferred to nest in waterways with tall (>60 cm) grass cover, whereas nest densities of vesper sparrows were greater in mowed waterways (Bryan and Best 1994). Grasshopper sparrows nested only in waterways that had been mowed the previous year; sedge wrens nested only in waterways that had not been mowed the previous year. The timing of mowing also may affect populations. Birds that have been displaced from mowed hayfields may move into grassed waterways with suitable vegetation structure. Mowing waterways at the peak or late in the nesting season may interfere with some birds' last nesting attempt of the season.

Terraces are another kind of strip cover that results from conservation-oriented land use in cropland. Wildlife use of terraces is poorly known but has been examined by D.W. Beck (USDA Soil Conservation Service, Des Moines,

Iowa, unpubl. data). Beck observed 13 bird species using grassed backslope terraces and found evidence of nesting by mallards, dickcissels, vesper sparrows, red-winged blackbirds, and ring-necked pheasants.

Fencerows are linear habitats that separate agricultural fields with a fenceline and associated vegetation. Best (1983) examined bird use of fencerows with only herbaceous vegetation, herbaceous vegetation with scattered woody plants, and continuous woody vegetation. The number of species was greatest in the continuous woody type and least in the herbaceous type, regardless of season.

In Michigan, 16 species nested in herbaceous and woody fencerows; nest density in fencerows was 43.5 nests/ha (Shalaway 1985). Fencerow width, adjacent field type, and area of open shrubs (i.e., <50 percent shrub cover 1.5-2.0 m above the ground) most influenced nest density. Wider fencerows, which were more heterogeneous and had greater shrub coverage than narrow fencerows, supported greater nest densities. Fencerows bordered by old fields had more nests than those bordered by crop fields. Nest density and abundance increased with shrub abundance. Song sparrows, American robins, northern cardinals, red-winged blackbirds, gray catbirds, brown thrashers, northern flickers, and ring-necked pheasants were the most frequent nesters in fencerows. Apparent nest success (successful nests/active nests) in fencerows was 58 percent overall. Raccoons (*Procyon lotor*), red foxes (*Vulpes vulpes*), striped skunks (*Mephitis mephitis*), and long-tailed weasels (*Mustela frenata*) were responsible for most nest losses. Nest success was lower for larger, ground-nesting game birds than for passerines nesting on or above the ground.

Herbaceous cover along roadsides is used by many species. In Iowa, 35 bird species were observed in roadsides compared with 26 species in adjacent rowcrop fields (Camp and Best 1993). Grass was the dominant vegetation in roadsides and forbs were uncommon. Increased diversity of roadside vegetation can increase bird species richness (Paruk 1990, Warner 1992). Abundance of some bird species in roadsides was related to vegetation height and vertical density (Camp and Best 1993). Both of these vegetation characteristics are influenced by burning and the composition of grasses (exotic vs. native). The number of bird species observed was inversely related to grass coverage and directly related to the amount of bare ground (Camp and Best 1993). Red-winged blackbird nests were the most common, and nest success (Mayfield estimate) was 26 percent. Predation was the main cause of nest failure, accounting for the fate of 52 percent of all active nests. Mowing appeared to benefit some species; vesper sparrows and meadowlarks nested on mowed roadside shoulders.

In Illinois roadsides, 92 percent of all passerine nests were of red-winged blackbirds; other nesting species included dickcissels, brown thrashers, eastern meadowlarks, grasshopper sparrows, vesper sparrows, song sparrows, and sedge wrens (Warner 1992). Nest densities in roadsides were affected by the kind of habitat in the vicinity of the roadside. The number of passerine nests in roadsides was higher in years in which a smaller proportion of the study area was planted to small grains, presumably because less of the non-roadside habitat was suitable for nesting. Within years, the 4-mi^2 study plot with the most grassland area had the highest density of passerine nests in linear, grassland habitats. Similarly, roadside study plots with the highest density of pheasant nests were near other prime nesting habitats such as hay (Warner and Joselyn 1986, Warner *et al.* 1987). In Iowa, the highest densities of red-winged blackbirds were found in roadsides adjacent to idle grasslands and hayfields (M. K. Koob and L. B. Best, Iowa State Univ., Ames, unpubl. data).

Densities of passerine nests in Illinois roadsides also were affected by characteristics of the roadsides (Warner 1992). Nest densities were greater in interstate roadsides than in secondary roadsides and increased with width of the roadside. Nest densities were four to five times greater in interstate roadsides dominated by brome-alfalfa (*Bromus inermis-Medicago sativa*) than in those dominated by fescue (*Festuca* spp.). There were over six times more passerine nests on managed (mowing deferred until 1 August) than on unmanaged (frequently mowed) secondary roadsides. In this Illinois study, as in the Iowa study (Camp and Best 1993), mowing was confined to the roadside shoulders; the sloped sections were left unmowed.

Higher nest densities in a particular landscape do not necessarily indicate that a species is doing better there; high densities could lead to lower nest success. Ring-necked pheasants in Illinois had low annual nest success in linear habitats (farmland corridors and managed roadsides) in years with low amounts of grassland (strip-cover habitats, forage crops, small grains) per hen in the spring (Warner 1994). Also, nest success in roadsides was lower in years with high nest densities than in years with low densities (Warner *et al.* 1987). Similar findings have not been reported for passerines.

Shelterbelts are a common feature of midwestern landscapes (Johnson and Beck 1988). For many decades, federal agencies have encouraged farmers to plant shelterbelts or windbreaks to decrease wind erosion and to protect crops, livestock, and farmsteads. Establishment of shelterbelts can provide positive economic returns (Brandle *et al.* 1992). Shelterbelts usually consist of one to five rows of trees and shrubs (Capel 1988) but can be much wider, especially around farmsteads. Because shelterbelts usually contain trees, they provide habitat for forest and forest-edge species.

In North Dakota, at least 64 species of birds are known to have bred in shelterbelts or tree claims (trees planted by homesteaders) (Cassel and Wiehe 1980). The most common species were the brown thrasher, mourning dove, vesper sparrow, least flycatcher, eastern kingbird, black-billed cuckoo, yellow warbler, American goldfinch, gray catbird, clay-colored sparrow, and American robin. The vesper sparrows and American goldfinches were presumably not nesting in the shelterbelts (see Yahner 1982). Yahner (1982, 1983a) recorded 87 bird species using farmstead shelterbelts in Minnesota and documented nesting in 17 of these. In South Dakota, the number of bird species increased with shelterbelt area both during spring migration and during the breeding season (Martin 1980, 1981). The density and diversity of breeding birds was significantly correlated with both the age and size of shelterbelts in North Dakota (Cassel and Wiehe 1980).

The configuration of shelterbelts can affect the species composition of the avifauna. Shelterbelts with only a few rows of woody plants attracted more birds associated with open habitats, whereas those with many rows attracted more birds associated with forested habitats (Cassel and Wiehe 1980). Yahner (1983a), however, provided correlations between variables describing vegetation structure and avian community or population variables in shelterbelts and found that the perimeter and length of shelterbelts were associated more often with measures of community structure than were area and width of shelterbelts.

The avian species composition of shelterbelts also can be affected by vegetation structure and landscape context. Martin and Vohs (1978) found the highest bird species diversity in South Dakota shelterbelts with developed tree canopies and lush grass layers; dense shrub growth under the trees was not preferred. In Minnesota, vegetation structure was a major factor determining bird community structure in shelterbelts, with older belts having lower densities of shrubs, higher densities of trees, and greater bird species richness (Yahner 1983a). Distance to wooded and old-field habitats, and the amount of cropland and pastureland surrounding shelterbelts, influenced which birds were found in shelterbelts. A concentration effect occurred in isolated shelterbelts, which were more likely to have certain species, probably because of the paucity of similar habitat in the vicinity. In addition to species occurrence, it is important to examine nest success (Yahner 1982, 1983b). Apparent nest success (successful nests/active nests) of mourning doves and American robins in shelterbelts was 32 and 56 percent, respectively.

Additional perspectives on shelterbelts can be found in several papers. Johnson and Beck (1988) reviewed shelterbelt management and wildlife. They classified bird species on the basis of how much they benefit from shelterbelts and discussed characteristics of shelterbelts that relate to wildlife use. Schroeder (1986) presented a Habitat Suitability Index model for wildlife species richness in shelterbelts which reliably predicts species richness over much of the Great Plains (Schroeder *et al.* 1992). Podoll (1979) distinguished between shelterbelts as a habitat that is essential for some species but merely used by other species. Johnson (1996) points out that trees may have detrimental effects on prairie birds.

CONSERVATION STRATEGIES

This review suggests that current agricultural practices are contributing to population declines of farmland birds, although not necessarily to the exclusion of factors acting during migration and on the wintering grounds. Other investigators have reached similar conclusions (e.g., Rodenhouse *et al.* 1993, 1995; Herkert 1994, 1995; Warner 1994). Recent population declines could be due mainly to loss of habitat (Rodenhouse *et al.* 1995) or to a combination of loss of habitat and degradation of habitat (e.g., low nest success or survival rates in attractive habitats and landscapes). Conservation measures to deal with these declines will depend on the relative importance of habitat loss and degradation, the objectives of a conservation strategy, the costs of implementing a strategy, and the feasibility of the strategy.

Owners and operators of farms can implement agricultural practices that will benefit NTMBs on midwestern farmland. Many farmers appreciate having non-pest bird species on their farms and nearby. If farmers are forced to choose, however, between economic survival and having wildlife to observe and hunt, they will choose survival. Even in situations in which economic survival is not at stake, farmers will consider costs. It is in these situations where information and education (e.g., university extension) may make the difference in favor of wildlife generally and NTMBs specifically. Here we take the practical approach of recognizing that recommendations for farmland must be aimed at a broad audience that includes private landowners as well as professional wildlife managers. We offer three objectives that we believe will have broad support, review a general framework that likewise should have broad support, and summarize specific recommended modifications to current agricultural practices.

Objectives

A primary objective of a conservation strategy for NTMBs in midwestern agricultural landscapes should be to conserve the remnants of prairie and savannah that still exist. Most of the prairie has been lost (Samson and Knopf 1994), increasing the value of the remnants. Although restoration of midwestern prairie and savannah ecosystems on a large scale is impractical, the remnants can be used for biological investigations, ecological education, and core areas for additional restoration. Although the persistence of grassland and open-woodland birds in the Midwest may not depend on these remnants, it is important for the conservation of biodiversity in general to save or restore representative natural ecosystems with a large proportion of their native flora and fauna. Of similar importance are the remnant woodlands, riparian areas, and wetlands required by a variety of bird species (Best *et al.* 1995, table 1).

A second objective of a conservation strategy should be to ensure the population viability of species that are (1) listed under the Endangered Species Act (ESA), (2) listed by individual states, or (3) are of special concern to managers because their populations are declining or vulnerable (USFWS 1995). Forty-one of these species occur in midwestern farmland (table 1). Investments that could prevent listing of additional species under the ESA may be economical in the long run, given the costs of listing, preparing recovery plans, etc. Beyond this practical reason, drastic population declines of rare species will result in loss of these species from some ecosystems, which may affect ecosystem functioning and be undesirable for other reasons (Wiens and Dyer 1975, Ehrlich and Mooney 1983, Saunders *et al.* 1991, Chapin *et al.* 1992, Baskin 1994).

A third objective should be to identify priority species or groups of species beyond those identified under the second objective. A different conservation strategy for these NTMBs might be called for, depending on whether the primary concern is for reversing population declines or for promoting biodiversity. Some species may require large blocks of grassland habitat (Herkert 1994); others may benefit from increased availability of edge habitat in the form of strip cover. Ideally, management can benefit all groups (Tome *et al.* 1994).

General Framework

Identification of sources and sinks (Wiens and Rotenberry 1981, Pulliam 1988) is an important step in implementing a conservation strategy for NTMBs in farmland (Rodenhouse *et al.* 1993). This step has been taken for forested habitats in the Midwest (Donovan *et al.* 1995). Many managers now understand that creating attractive habitat may not necessarily benefit target species if that new habitat is a sink for

those species. The review by Rodenhouse *et al.* (1993) of nest success in farmland indicates that few species are known to be reproducing at levels sufficient to balance estimated mortality. Even the nest success estimates in CRP fields and moderately grazed pastures, reviewed in the "Idling cropland" section, are not encouraging.

After sources and sinks have been identified, habitat, land-use, and landscape features that distinguish them can be examined to determine which features contribute to differences in vital rates. Such analyses can lead to recommended practices that will change the attractiveness of sources and sinks or increase individual fitness in them. Simulation modeling can be useful in evaluating potential actions (e.g., Cowardin *et al.* 1983). Finally, after actions have been taken, the status and trends of target species or guilds should be monitored to evaluate the results. All of this can be done in the context of a comprehensive framework for conservation and management planning (Freemark *et al.* 1993, 1995). A major limitation of this process is our incomplete understanding of factors affecting population dynamics and ecosystem functions in agricultural ecosystems.

Specific Recommendations

Rodenhouse *et al.* (1993) made the insightful observation that farming might develop along two tracks. Small, owner-operated farms would predominate close to urban centers where farm families could obtain supplemental income. These farms would provide habitat for many NTMBs because expensive inputs (large machinery, fuel, inorganic fertilizer, and pesticides) would be minimized and farming would be diversified. Large, externally owned farms would exist in rural areas with the most productive soil. These would be intensively managed and provide little non-crop habitat. These tracks certainly seem reasonable, considering current economic conditions in the Midwest.

Agricultural practices on larger farms may best be influenced by modifying agricultural programs and policies. These farms can be expected to attract species nesting in cultivated fields: horned larks, vesper sparrows, and killdeer. Although these species are not common in cropland (table 1), their cropland habitat is very extensive. Populations of these species are large. Monitoring of population trends of these species should continue. As long as their populations are relatively stable (Johnson and Schwartz 1993b), these species should not receive high conservation priority. Vesper sparrow populations have declined recently in parts of the Midwest, according to data from the North American Breeding Bird Survey, and have low nest success in Iowa (Perritt and Best 1989).

Smaller farms should be encouraged to enhance the attractiveness of their land to NTMBs, especially grassland birds that have declined in recent decades, such as grasshopper sparrows and bobolinks. Long-term setaside, such as the CRP, provide much of the grassland habitat in the Midwest today and should be encouraged. A wide variety of other species, including woodland species, also can benefit from farmland (table 1). The following recommendations are organized by agricultural practice, as covered above.

Cultivation.—Tillage should be minimized. Rodenhouse *et al.* (1993) recommended that crop residue be retained on the soil surface, which could be done by reducing the number of times a field is cultivated annually and by using subsurface tillage where appropriate. A greater variety of NTMBs may be attracted by increasing the diversity of crops grown, especially if the structure of the vegetation is quite different. Crop diversity also will contribute to sustainable agriculture (Barrett *et al.* 1990).

Grazing.—Grazing is not incompatible with a diverse avifauna. Moderate grazing of pastures may enhance local habitat diversity. It has been suggested, however, that public grasslands managed for wildlife should benefit species that prefer habitat that is in relatively short supply; that is, grassland that is not frequently grazed or hayed (Kirsch *et al.* 1978). This recommendation could be extended to private land in the Midwest. Each grassland species has a particular kind of preferred vegetation for nesting (Owens and Myres 1973, Skinner *et al.* 1984), facilitating management for particular groups of species. Further grazing recommendations are in Herkert *et al.* (1996).

Haying.—Haying appears to be a major problem for NTMBs because hayfields have low nest success (Frawley 1989, Bollinger *et al.* 1990). They are highly attractive to certain

species, but the cutting interval is usually too short to allow complete nesting cycles. Rodenhouse *et al.* (1993) recommended that spring mowing be delayed as long as possible, nighttime mowing be avoided, and the intervals between mowings be as long as possible. Additionally, hayfields with warm-season grasses should be encouraged because they would be cut later in the year than alfalfa. Further recommendations are in Herkert *et al.* (1996).

Idling cropland.—Idled cropland would seem to require relatively little management because it already attracts a variety of species. Practices such as mowing or burning, however, could modify many areas and change the composition of the avifauna. Mowing during the nesting season can lower nest success substantially. It may be possible to enhance bird reproduction and survival by changing the amount and configuration of idle land; nest success may be higher in block habitats than in linear ones with proportionately more edge habitat. In a Minnesota study, nest success in tall-grass-prairie remnants was lower near woody edges than it was farther from such edges (Johnson and Temple 1990). Finally, if certain kinds or configurations of idled cropland are found to be source habitats, they should be strongly encouraged to counteract the many apparent sinks in agricultural landscapes.

Patterson and Best (1996) examined bird use of CRP fields that differed in structure and composition of vegetation and made recommendations aimed at benefiting grassland birds. The species composition of the vegetation should be based in part on vegetational attributes that attract birds. A wider variety of planted species than has been used in the past would enhance bird use. Mowing of weeds should be done after the peak of the nesting season. Some large tracts of habitat should be maintained for the benefit of area-sensitive species. Other recommendations could be added to their list. Grazing may enhance species richness if the structural diversity of the habitat is increased. Prescribed burning, every few years, is often recommended for other grassland habitats to maintain attractive nesting cover and presumably would benefit birds using CRP fields. Multiple-year programs, such as the CRP, appear to offer many more benefits to NTMBs than annual set-aside programs. Grassland cover that is at least a year old is established well enough in the spring, when migratory birds arrive, to attract breeding pairs. After a few years, vegetative and sexual reproduction by plants fills in gaps from the initial seeding, increasing the height and density of vegetative cover.

Maintaining strip cover.—Recommendations for strip cover depend on whether management is intended to favor grassland species or edge species. Management for the former should favor herbaceous cover. The benefits to NTMBs of permanent herbaceous cover can be enhanced by changing common practices. Timing of mowing of grassed waterways should be considered. Because peak nesting in grassed waterways occurred in July in Iowa, Bryan and Best (1991) recommended that mowing in waterways be deferred until the end of August or early September. Fall mowing is not recommended because residual cover would be reduced in the subsequent winter and spring, and annual mowing is discouraged (Bryan and Best 1994).

Beck (unpubl. data) found several factors that decreased wildlife use of grassed-backslope terraces, such as steeper backslopes, poor grass stands, narrow width, grazing, and herbicide drift. He suggested that improved management might involve flattening backslopes, improving grass stands, improving grazing management, and planting alternative vegetation.

If herbaceous vegetation on roadsides is mowed, it should be mowed in mid-to-late August (Camp and Best 1993). Also recommended is prescribed burning in the spring every 3 to 5 years. Both mowing and burning should be conducted in blocks to ensure that some portions of roadsides are undisturbed at any time. Fencelines should be retained along roadsides because roadsides with fencelines are less susceptible to agricultural encroachment and fencelines provide singing perches for birds. Mowing roadside shoulders provides nesting habitat for some species, but mowing should be restricted to early spring and late summer to reduce nest losses.

An adoption model for roadside (and other) habitat management involves farm operator attitudes and perceptions (Warner 1983). Warner (1992) made the following roadside management recommendations for grassland

birds: (1) fescue and bluegrass (*Poa*) sods should be seeded to brome-alfalfa and/or native grasses and forbs, (2) mowing should be delayed until after 1 August, (3) widths of roadside tracts should be maximized where possible, and (4) the establishment and protection of woody plants should be encouraged. In areas where brown-headed cowbirds are abundant, the effect of woody plants on frequency of brood parasitism will have to be considered carefully. The prescribed burning recommended by Camp and Best (1993), which would slow the establishment of woody plants, could be used in areas where woody plants were not desired. They also noted that bird use of roadsides with well-established native vegetation has not been evaluated. Such roadsides may become more prevalent in the future as the agencies charged with managing roadside vegetation seek methods that will benefit wildlife and minimize invasion of weeds.

Managing for breeding, edge species should be focused on providing diverse habitat in which nest success can potentially be high. Fencerows should be >3m wide, according to the recommendation of Shalaway (1985), who worked in Michigan. This would allow for greater shrub/tree growth and potentially reduce the likelihood of nest predation. Farther west, where brown-headed cowbirds are more abundant, the possible enhancement of habitat for this species would have to be considered. Shalaway (1985) also suggested that selective mowing and woodcutting in fencerows can create a mosaic of vegetation types attractive to a variety of birds. He also recommended retaining or creating at least 1 to 2 large snags per km of fencerow. Artificial nest-boxes may attract eastern bluebirds and tree swallows (Robertson *et al.* 1992), both NTMBs.

Management recommendations for shelterbelts, given by Yahner (1983a), include: establish shelterbelts at least 0.6 ha in area and 8 rows in width, consider proper spacing within and between rows, use a diversity of plantings, discourage livestock grazing, discontinue mowing and cultivation after the shelterbelt becomes established, and retain snags as nesting and foraging sites.

General.—Our final recommendation is that biologists and managers strive to improve their working relationships with private landowners and the organizations that represent them. The concept of benefiting NTMBs in agricultural landscapes is different from much of traditional wildlife management, which has focused on managing land controlled by state and federal governments. Those seeking to benefit NTMBs can learn from those who have sought to benefit ducks and upland game species, most of which occur on private land. Many game managers have learned the value of education, outreach, and attention to landowner concerns. Progress may require compromise and consideration of large-scale solutions, such as influencing agricultural policy. The recent efforts to renew the CRP brought together many wildlife and farming organizations that shared a common interest. Serious consideration has been given to the major effects of agricultural policies on wildlife in agricultural landscapes (Risley *et al.* 1995). With enhanced communication, sound science, and a common conservation strategy, we are confident that farmers and wildlife can continue to coexist in the Midwest.

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MANAGEMENT OF MIDWESTERN GRASSLAND LANDSCAPES FOR THE CONSERVATION OF MIGRATORY BIRDS

James R. Herkert[1], David W. Sample[2], and Richard E. Warner[3]

ABSTRACT.—Declines of Midwestern grassland birds and habitats point to the need for heightened conservation attention for grassland habitats and their associated breeding bird species. We review the conservation and management of migratory birds in Midwestern grasslands to develop management recommendations for the conservation of breeding birds in these landscapes. We used a priority ranking system to identify species of greatest management concern in the region. The Henslow's sparrow was identified as the species of highest management concern. Highly ranked bird species were associated with a variety of grassland habitats and habitat structures, including dry prairies, pastures, old fields, hayfields, wet prairies, sedge meadows and grasslands with interspersed shrubs. The diverse habitat associations of bird species with high regional management concern suggests that the problems facing Midwestern grassland birds are widespread and involved a wide variety of the region's grassland habitats. One common feature among many of the highly ranked grassland bird species was a sensitivity to habitat fragmentation, suggesting that this may be a general problem facing grassland birds in the region.

Declines in Midwestern grassland bird numbers were significantly correlated with declines in the regional acreage of pastures and hay fields. Other major landscape scale phenomenon likely impacting grassland birds in the region include habitat fragmentation and ill-timed cutting of hay fields. At a local scale, grassland birds have a variable response to management with some species being most common on grasslands recently disturbed with prescribed burning or grazing while others are most common in undisturbed areas. Grassland bird nest success can also be significantly influenced by grassland management, with nest success tending to be highest in the 2-3 years following prescribed fire. As a result of the of the variable response of grassland birds to prescribed fire, a mid-length (3-5 year) rotational burn program appears to be optimum under most circumstances. In grazed systems, a rotational system is also most desirable with a majority of areas being light to moderately grazed.

There is evidence that declines in the availability of grassland habitat in the region may be significantly influencing regional grassland bird declines. There is also some evidence that grassland bird nest success in many areas is below levels believed necessary for population maintenance. Limited winter habitat and/or winter resources have also been implicated in the declines of some grassland bird species. Efforts to identify which factors are most important in limiting grassland bird populations are hampered by limited data on many aspects of their ecology. More data on grassland bird demographics are needed in order to identify and differentiate habitats that are sources from those that are sinks, and a greater understanding of the winter ecology for most grassland bird species is also needed.

[1]*Illinois Endangered Species Protection Board, 524 South Second Street, Springfield, Illinois 62701.*

[2]*Wisconsin Department of Natural Resources, 1350 Femrite Drive, Monona, Wisconsin 53716.*

[3]*Department of Natural Resources and Environmental Studies, University of Illinois, 213 Mumford Hall, Urbana, Illinois 61801.*

INTRODUCTION

Populations of many species of grassland birds have declined significantly over the last 30 years (Peterjohn *et al.* 1994). Analyses of Midwestern breeding bird population trends show that grassland birds have declined more extensively than have birds associated with other habitats in the region (Herkert 1995). In fact, four of the five fastest declining species in the region (grasshopper sparrow, western meadowlark, bobolink, and loggerhead shrike) are associated with grassland habitats. Recent analyses of continental breeding bird population trends have also shown that the declines exhibited by grassland birds have been in general steeper, more consistent, and more widespread than declines in other groups of North American birds (Knopf 1994). Native grasslands throughout North America have also suffered substantial declines, with these declines being particularly severe in the Midwestern United States (Noss *et al.* 1995). Mesic tallgrass prairie, sedge meadow, and Lakeplain wet prairie are three Midwestern grassland ecosystems that have recently been identified as critically endangered in the United States (Noss *et al.* 1995). These declines point to the need for heightened conservation attention for grassland habitats and their associated breeding birds.

We review issues relevant to the conservation and management of migratory non-game grassland birds and their habitats in Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio and Wisconsin. Both short- and long-distance migrants are included, but we do not consider resident species. Short-distance migrants deserve management attention because many have experienced long-term population declines as great as or greater than many neotropical migrants (Peterjohn *et al.* 1994, Herkert 1995). We focus on two major grassland habitats: those habitats that are typically undisturbed during the breeding season, both native and non-native, including prairies, sedge meadows, prairie restorations, oldfields, fallow fields, and cool- and warm-season grass fields planted for wildlife cover; and agricultural habitats that form perennial sods but which are disturbed during the breeding season, such as hayfields and pastures. Other agricultural and grassland habitats are considered elsewhere (see Koford and Best 1996, Johnson 1996).

OVERVIEW OF GRASSLAND HABITATS IMPORTANT FOR GRASSLAND BIRD CONSERVATION IN THE MIDWEST

Native Prairies

Approximately 162 million ha of prairie occurred in North America before extensive European settlement; native prairie was the largest vegetation province in North America (Samson and Knopf 1994). Traditionally, prairies in North America have been divided into three subcategories; an eastern tallgrass section, a central mixedgrass section, and a western shortgrass component. Of the original 68 million ha of tallgrass prairie in North America, approximately 57 percent (38 million ha) occurred in the Midwestern states considered here. Declines in acreage of tallgrass prairie exceeds those reported for any other major ecosystem in North America (Samson and Knopf 1994). In the Midwest <1 percent of the region's native prairie remains intact (Samson and Knopf 1994). In many Midwestern areas, all that remains of once vast stretches of prairie are small, isolated remnants (e.g., Smith 1981, Schwegman 1983). For example in Illinois, <20 percent of the state's 245 native prairie remnants are >10 ha, and <4 percent are larger than 40 ha (Herkert 1994a). Currently, most native prairie in the region occurs in Minnesota and Missouri; both have more than 30,000 ha despite declines of >95 percent (Samson and Knopf 1994).

Large native prairies in the Midwest can support relatively diverse bird species assemblages (e.g., Sample 1989, Herkert 1991a). However, the small size of most Midwestern prairie remnants limits their attractiveness for many grassland birds (e.g., Herkert 1994a). Breeding bird communities on Midwestern prairies are dependent not only on tract size, but also on soil moisture, habitat structure, degree of woody invasion and plant species composition (e.g., Hoffman and Sample 1988, Sample and Hoffman 1989, Mossman and Sample 1990). For example, grasshopper, lark, and vesper sparrows are most common in dry prairies in the Midwest and are either uncommon or absent on wet or wet-mesic prairies, whereas sedge wrens and bobolinks are much more common on wet prairie sites than on dry prairies (Hoffman and Sample 1988, Sample and Hoffman 1989).

Other native communities related to prairies also provide valuable habitats for some grassland birds. For example, in Wisconsin sand prairies and barrens are important for lark and vesper sparrows, and large expanses of open barrens are important for upland sandpipers (Mossman *et al.* 1991).

Conservation of native grassland communities on unplowed prairie sod has the important additional conservation benefit of preserving a broader spectrum of the overall biotic diversity of grassland ecosystems—including plant species, insect populations, and soil microfauna and microflora—than do exotic grasslands on plowed soil (Henderson and Sample 1995).

Sedge Meadows

Sedge meadows are wet "grassy" meadows that are dominated by sedges (*Carex* spp.), often containing a significant grass component (Mossman and Sample 1990). Estimates of the original or current acreage of sedge meadows in the Midwest are uncertain, but states such as Wisconsin, Minnesota, Iowa and Illinois probably had the most. Like the tallgrass prairie, sedge meadows have been much reduced in the Midwest and most are highly fragmented. For example, of an estimated 500,000 original ha of sedge meadows in Wisconsin, only about 12,500 ha of high-quality meadow remains (Mossman and Sample 1990). Due to their unique hydrology, structure, and flora, sedge meadows provide the primary habitat for some distinctive grassland birds including sedge wrens and LeConte's sparrow even though overall sedge meadows do not support highly diverse bird communities. In Wisconsin, LeConte's sparrow and several other wetland specialist species occur in almost no other grassland habitat than sedge meadow (Mossman and Sample 1990). Many other wetland habitats also are used by grassland birds; in Wisconsin these include shrub-swamps and open bogs (Mossman and Sample 1990).

Restored Prairies

Restored prairies are grassland areas with diverse histories that have been planted with native grasses and forbs in an attempt to recreate prairie in areas where native prairie habitats have been lost. Restored prairies in the Midwest tend to be small and are frequently dominated by dense growths of prairie grasses, but some restorations possess a diverse forb component. The small size of many Midwestern grassland restorations may limit their conservation benefit, since most lack several grassland bird species of greatest conservation concern in the region (e.g., Herkert 1991b). Prairie restorations that are heterogenous and fairly large in size can be effective in attracting sizable populations of grassland birds (e.g., Sample 1989, Volkert 1992). Related to prairie restorations are planted stands of native, warm-season grasses, which often occur on public lands (wildlife areas, Waterfowl Production Areas, etc.) managed for gamebird production. These fields tend to be grass-dominated and have few native forbs, but weed forbs and grasses are often present on poorly established fields. Grassland bird use of these fields is variable. Relatively few species use fields with monotypic grass cover. Some of these birds, however, are specialists that only occur in other tall dense habitats such as sedge meadows. Fields that are weedy and less dense usually harbor a greater diversity of species (Sample 1989).

Hayfields and Pastures

Hayfields and pastures currently have extensive acreage. These habitats have been important in sustaining populations of some Midwestern grassland bird species as native prairies were lost. In addition, expansion of pastures and hayfields into portions of the Midwest that had previously been mostly forested allowed some grassland bird populations to greatly expand their range in some areas of the Midwest (e.g., Mayfield 1988, Brewer *et al.* 1991). This range expansion helped partially offset the tremendous loss of native prairie habitat in the prairie sections of the region and probably helped stabilize regional populations for those species that were successful in colonizing these new grassland habitats.

Early hayfields in the prairie sections of the region tended to be comprised primarily of native grasses (e.g., Warner 1994) and therefore presumably supported a grassland bird fauna that was largely similar to the native prairies although some species adapted to relatively short grass structure and periodic

disturbance may have increased in abundance in these fields. Early in this century, these native grasses were replaced by non-native grasses (Warner 1994), and more recently the non-native grasses have been largely replaced with varieties of alfalfa hay. In 1992, alfalfa hay accounted for 53 percent of the 6.2 million ha of hay in the region (U.S. Department of Commerce 1994). In the 1950's hayfields continued to provide an important habitat for Midwestern birds. Graber and Graber (1963) estimated that even though hayfields comprised only 7 percent of Illinois landscape in the late 1950's, as much as 20 percent of the entire breeding bird fauna in the state may have resided in this habitat.

Pastures are currently the region's most abundant grassland habitat, accounting for 8.7 million ha in the Midwest (roughly 7.5 percent of the region's land area, U.S. Department of Commerce 1994). Pastures also were important refuges for grassland birds as prairies were converted for agriculture (e.g., Ridgway 1895, Forbes 1908, Graber and Graber 1963). Pastures are still important grassland bird habitats because, when not overgrazed, they can support diverse assemblages of grassland bird species including many species with declining populations (e.g., Sample 1989).

Pastures and hayfields reached their peak abundance in the Midwest in the early 1900's and have declined in acreage almost continuously ever since (fig. 1). Over the last 50 years, combined pasture and hayfield acreage has declined by more than 50 percent (~17.0 million ha) in the Midwest and is now at its lowest level in more than 100 years (fig. 1).

Old Fields, Introduced Cool-season Grasses, and Other Non-prairie Grasslands

There are several other types of grassland habitats in the Midwest that are important for grassland birds. Old fields are attractive to many Midwestern grassland birds including many species of high conservation concern (Sample 1989). Without periodic disturbance, old fields can quickly succeed to brushy shrublands that are generally of minimal conservation value to grassland bird species. Fallow fields (fields plowed but not planted in the current or previous year) also provide some habitat for grassland birds in the Midwest. Grassland bird use of fallow fields is generally lower than old fields and most other types of grassland habitats (see Sample 1989). Estimates of the area of the various types of miscellaneous grasslands in the Midwest are scant. In 1992 more than four million ha of cropland was idle, fallow, or in cover crops, legumes, or soil improvement grasses in the Midwest (U.S. Department of Commerce 1994), suggesting that these miscellaneous grasslands may provide significant habitat for grassland bird populations in the region.

Fields of introduced, cool-season grasses, both monotypic and grass-legume mixtures, such as state wildlife areas and Waterfowl Production Areas are locally common in parts of the Midwest region. These habitats are important to some non-game grassland bird species; in Wisconsin these are often species that commonly nest in hay, such as bobolinks (Petersen *et al.* 1982, Sample 1989).

Other miscellaneous grassland bird habitats are young conifer plantations (field, vesper, and clay-colored sparrows), orchards (loggerhead shrikes), and retired pasture; these habitats may be of local conservation importance. Grassed roadsides and waterways, long-term farmland set-aside fields, and other crop fields are other examples of grassland habitats in the Midwest. Koford and Best (1996) discuss the value of these habitats for migratory birds.

STATUS OF MIGRATORY GRASSLAND BIRDS IN THE MIDWEST

Of the 26 species of grassland birds that breed in the Midwest region, 8 are neotropical migrants (spend their nonbreeding period primarily south of the United States), 14 are short-distance migrants (winter extensively in North America, although some populations winter south of the United States) and 4 are non-migratory, resident species. Only the 22 long- and short-distance grassland migrant species are considered here. In addition to the migratory grassland bird species, we have also included five species of migratory "shrubland" birds that require an open shrub/grass matrix and are of conservation concern in the Midwest in our assessment (loggerhead shrike, Bell's vireo, clay-colored sparrow, lark sparrow, field sparrow).

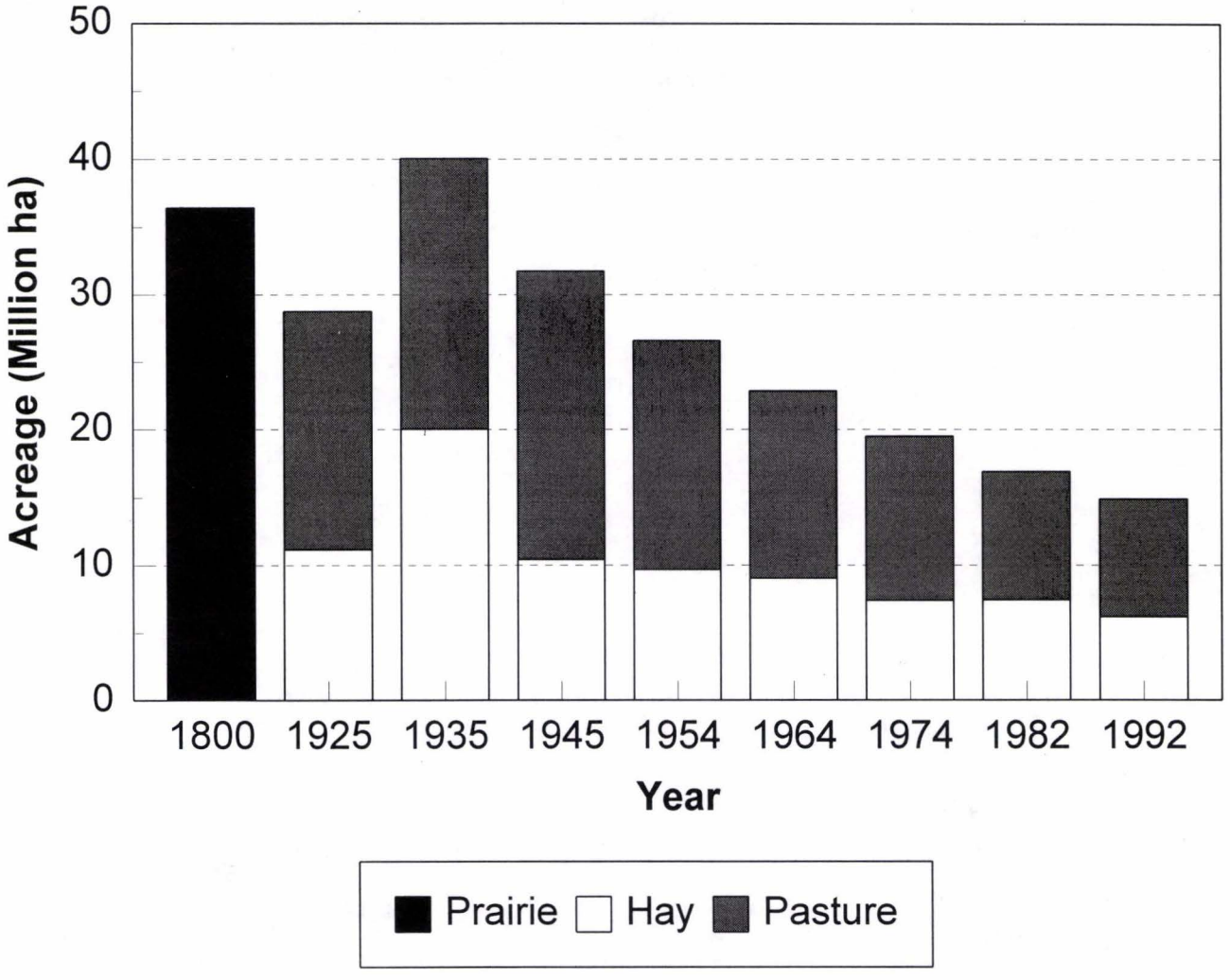

Figure 1.—*Acreage of major grassland habitats in the Midwestern United States 1800-1992. Figures for prairie were obtained from Samson and Knopf 1994, figures for hay and pasture were obtained from the U.S. Census of Agriculture (e.g., U.S. Department of Commerce 1994).*

Population Status and Trends of Migratory Grassland Birds

Populations of 10 species of Midwestern grassland birds have shown significant regional population declines over the last quarter century based on federal Breeding Bird Survey data (table 1). Only one migratory grassland bird species (killdeer) has shown a significant regional population increase over this same period. Many of these grassland birds are also undergoing significant population declines at the national level (table 1). Additionally, many migratory grassland bird species have very small populations in some portions of the region and as a result are considered to be either threatened or endangered in some of the Midwestern states in which they occur. Nearly half (48 percent) of the 27 grassland-associated species we examined are considered to be endangered or threatened in at least one Midwestern state (table 2).

Species Priority Ranking

We ranked our grassland-associated species according to regional management priority using a modification of the Partners in Flight priority ranking scheme (Hunter *et al.* 1993, Thompson *et al.* 1993). To identify grassland birds of greatest management concern in the

Table 1.—*Breeding distribution and population trends for migratory bird species associated with Midwestern grassland habitats.*

Species	# Midwestern states in which species breeds	Breeding Bird Survey population trend - 1966-1994	
		Midwest[1]	U.S.[1]
Northern harrier	8	-2.0	-1.2*
Killdeer	8	2.2***	0.1
Upland sandpiper	8	-0.6	2.0***
Common barn owl	8	*a*	*a*
Burrowing owl	3	*a*	*a*
Short-eared owl	8	*a*	1.9
Common nighthawk	8	0.3	-0.9
Horned lark	8	-0.4	-0.8**
Sedge wren	8	1.7	1.7*
Sprague's pipit	1	*a*	-0.1
Loggerhead shrike	8	-8.2***	-3.5***
Bell's vireo	8	4.3	-3.0**
Dickcissel	8	-3.6***	-1.6***
Clay-colored sparrow	5	-0.6	-1.3
Field sparrow	8	-3.0***	-3.4***
Vesper sparrow	8	-1.7***	-0.9**
Lark sparrow	7	-2.7	-3.4***
Lark bunting	2	*a*	-0.7
Savannah sparrow	7	-1.1**	-1.2***
Baird's sparrow	1	*a*	-0.9
Grasshopper sparrow	8	-5.5***	-3.7***
Henslow's sparrow	8	-7.6***	-8.2***
Le Conte's sparrow	3	1.3	-0.7
Chestnut-collared longspur	1	*a*	-0.3
Bobolink	8	-3.3***	-1.8***
Eastern meadowlark	8	-2.9***	-2.6***
Western meadowlark	8	-4.0***	-0.6*

[1] * = $p < 0.10$; ** = $p < 0.05$; *** = $p < 0.01$; *a* = too rare for BBS trend estimation.

region, we used estimates of species' global abundance, winter and breeding distributions, threats on the breeding and wintering grounds, and population trends, as well as an estimate of the importance of the Midwest to each species' North American breeding population (table 3). Ranks for species' global abundance, winter and breeding distributions, and threats on the breeding and wintering grounds for neotropical migrants were scored initially by Thompson *et al.* (1993) and Carter and Barker (1993); we added scores for some short-distance migrants. Ranks for the importance of the Midwest region for each species were estimated using field guide range maps (National Geographic Society 1983; Peterson 1980, 1990) and data from the BBS (VU-BBS computer program, Hines *et al.* 1994). Ranks for species' population trends (1966-1994) were based primarily on data from the Breeding Bird Survey. If BBS data were unavailable and we were unable to form an opinion regarding population trend for a species, we assigned a score of three for the population trend. In tabulating overall scores, we doubled the value for each species' importance of area score because we believed this was an important variable in our ranking system. We derived each species' overall score by calculating the geometric mean of each species' individual scores.

Most of the species we analyzed (55 percent) had mean management concern scores between two and three (table 3). Only three species had scores below two (common nighthawk, horned lark, and killdeer). Nine species

Table 2.—*Endangered and threatened birds associated with Midwestern grasslands*

Species	State Status[1]
Northern harrier	IL^{E} IN^{E} IA^{E} OH^{E} MO^{E} WI^{sc}
Upland sandpiper	IL^{E} IN^{E} MN^{SC} OH^{T} WI^{sc}
Common barn owl	IA^{E} IL^{E} IN^{E} MI^{E} OH^{E} WI^{E}
Burrowing owl	IA^{E} MN^{E}
Short-eared owl	IA^{E} IL^{E} IN^{E} MI^{E} MO^{E} MN^{SC} OH^{SC} WI^{sc}
Sedge wren	OH^{E} IN^{T} WI^{sc}
Sprague's pipit	MN^{E}
Loggerhead shrike[2]	IL^{T} IN^{E} WI^{E} MN^{T} MI^{E} OH^{E}
Bell's vireo[2]	WI^{T}
Baird's sparrow	MN^{E}
Henslow's sparrow	IL^{E} IA^{T} IN^{T} MN^{SC} OH^{SC} WI^{sc}
Lark sparrow[2]	OH^{E} WI^{sc}
Chestnut-collared longspur	MN^{E}

[1] E=endangered, T = Threatened, SC = special concern

[2] Associated with grasslands with scattered shrubs

had scores above three. The Henslow's sparrow was ranked as the grassland species of the highest management concern in the region, and was clearly separate from other highly ranked species. Interestingly, four of the five "shrubland" species were included among the region's ten highest-ranking grassland-associated species (table 3).

An important result of the ranking process was the observation that the ten highest ranking grassland-associated species were associated with a variety of grassland habitats and habitat structures. For example, grasshopper sparrows prefer habitats with relatively short-stature vegetation and diverse structure, including dry prairies, pastures, and old fields (Sample 1989); upland sandpipers also typically prefer areas that have relatively short cover (e.g., Skinner 1975, Ailes 1980, White 1983, Buhnerkempe and Westemeier 1988, Sample 1989), but in some areas they may also preferentially utilize large tracts of native grasslands such as open sandy barrens (e.g., Mossman *et al.* 1991); bobolinks prefer areas with lush grassy vegetation including hayfields, idle cool-season grasses, and low pastures and prairies (Sample 1989, Brewer *et al.* 1991); dickcissels prefer old fields, hayfields, and other idle grasslands with moderately tall herbaceous vegetation (Taber 1947, Gross 1921, Emlen and Wiens 1965, Zimmerman 1971, Brewer *et al.* 1991); Henslow's sparrows prefer undisturbed grasslands (native and non-native) that have tall, dense cover (Wiens 1969, Skinner *et al.* 1984, Zimmerman 1988, Sample 1989, Herkert 1994b); LeConte's sparrows and sedge wrens prefer very tall and dense vegetation, sometimes on upland sites but most often in wet prairies and sedge meadows (Mossman and Sample 1990); Bell's vireos, field sparrows and loggerhead shrikes require grasslands with interspersed shrubs (Walkinshaw 1968, Brooks and Temple 1990, Prescott and Collister 1993, Hands *et al.* 1989); and clay colored sparrows also prefer grasslands with interspersed shrubs, including cut-over forest lands and young pine plantations (Roberts 1936, Brewer *et al.* 1991, Mossman *et al.* 1991).

An examination of the diverse habitat associations of species with high regional management concern scores suggests that the problems facing Midwestern grassland birds are widespread and involve a wide variety of the region's grassland habitats. Many of the highest-ranking species, however, are also known to be sensitive to habitat fragmentation (e.g., Herkert 1994a, Vickery *et al.* 1994) suggesting that grassland fragmentation may be a common problem facing many of the region's grassland bird species of high management concern.

Table 3.—*Scores used to rank management concern for Midwestern migratory grassland bird species. A score of 5 indicates high management concern, 1 indicates low management concern. See Carter and Barker(1993), Hunter* et al. *(1993), and Thompson* et al. *(1993) for more details regarding the Partners In Flight ranking system.*

Species	GA[1]	WD[2]	WT[3]	BD[4]	BT[5]	IA[6]	PT[7]	TOTAL
Henslow's sparrow	4	3	3	4	4	5	5	4.05
Dickcissel	2	4	4	3	3	4	5	3.51
Bobolink	2	4	3	3	3	4	5	3.39
Sedge wren	4	4	3	3	4	5	1	3.31
Bell's vireo	3	4	3	3	3	3	4	3.22
Grasshopper sparrow	2	3	3	2	4	4	5	3.22
Upland sandpiper	3	3	3	3	4	3	3	3.11
Field sparrow	2	3	2	3	3	4	5	3.11
Clay-colored sparrow	3	4	2	4	2	3	4	3.02
Loggerhead shrike	3	3	4	2	3	2	5	2.85
Le Conte's sparrow	4	3	3	3	3	2	3	2.81
Common barn owl	4	2	3	3	3	2	3	2.67
Lark sparrow	3	3	3	2	3	2	4	2.67
Baird's sparrow	4	4	4	4	3	1	3	2.63
Chestnut-collared longspur	3	4	4	4	4	1	3	2.63
Vesper sparrow	3	2	2	2	2	3	5	2.61
Northern harrier	4	1	4	1	4	2	4	2.38
Sprague's pipit	3	3	3	4	3	1	3	2.36
Burrowing owl	4	3	3	3	3	1	3	2.36
Eastern meadowlark	1	1	3	3	3	3	5	2.35
Lark bunting	2	3	3	4	4	1	3	2.33
Savannah sparrow	2	2	3	1	3	2	5	2.28
Western meadowlark	1	2	3	2	3	2	5	2.28
Short-eared owl	4	1	4	1	4	2	2	2.18
Common nighthawk	2	2	2	1	2	2	3	1.93
Horned lark	1	1	1	1	1	2	5	1.45
Killdeer	1	1	1	1	1	2	1	1.19

[1]GA - Global Abundance, abundant or demonstrably secure = 1; common or apparently secure = 2; uncommon to fairly common, including locally common = 3; rare to uncommon = 4; very rare to rare = 5.

[2]WD - Winter Distribution, Very widespread = 1; widespread = 2; intermediate = 3; local = 4; very local = 5.

[3]WT - Severity of Threats on the Winter Grounds, no known threats = 1; minor threats = 2, moderate threats = 3; severe threats = 4; extirpation or extinction likely = 5.

[4]BD - Breeding Distribution, the area of the breeding range in North America. ≥76% of temperate North America = 1; 51-75% = 2; 26-50% = 3; 11-25% = 4; ≤ 10% = 5.

[5]BT - Severity of Threats of Breeding Grounds in Midwest Region (habitat loss and fragmentation, low nesting success, contaminants, human disturbance, etc.). No known threats = 1; Minor threats = 2; Moderate threats = 3; Severe threats = 4; extirpation or extinction likely = 5.

[6]IA - Importance of Midwest Region to Species. Less than 1% of population in region = 1; 1-10% = 2; 11-25% = 3; 26-50% = 4; > 50% = 5.

[7]PT - Population Trend in Midwest Region (based on Breeding Bird Survey data). Ranking is based on both regional and national population trends. Trends are shown as regional and national trends with + indicating a positive trend and - indicating a negative trend, an * indicates statistical significance (p<.10) for the trend estimate. Trends which were unknown at both the national and regional level were assigned a rank of 3.

		Regional Population Trend (1966-94)				
		+*	+	unk	-	-*
National	+*	1	1	2	3	3
Population	+	1	2	2	3	4
Trend	-	3	3	3	4	5
(1966-94)	-*	3	4	4	5	5

We agree with Millsap *et al.* (1990) and Hunter *et al.* (1993) who have stressed that priority ranking systems should not replace human judgement in the allocation of conservation resources, rather these systems should be used as one tool in determining where to direct resource conservation efforts. Moreover, Thompson *et al.* (1993) caution against using ranking systems to focus management on a limited number of highly ranked species and recommend landscape-level management aimed at addressing the needs of suites of highly ranked species. In light of the diverse habitat associations of our highly ranked species, our ranking exercise suggests that large scale, diverse grassland management is needed to meet the habitat needs of the migratory grassland bird species of the greatest conservation concern in the region. Regional variation both in the distribution and quality of grassland habitat and in the distribution and abundance of grassland birds means that bird species and habitats of highest management concern will vary from one part of the region to another.

Conservation issues for grassland birds of management concern

Close attention needs to be paid to the distribution and extent of grassland habitats and birds in the region to properly focus and maximize use of scarce management resources. Grassland habitats are not evenly distributed across the region (fig. 2), and tract size and quality vary regionally. The most efficient way to manage for particular habitats (e.g., pastures or sedge meadows) is to focus on areas where the most acreage currently is and where the best opportunities for management of large tracts or landscapes lie (Sample and Mossman, in press). For example, certain states and areas will be best-suited for maintenance of sedge meadows and sedge meadow birds (such as portions of Minnesota, Wisconsin, Michigan and Northern Illinois), while others will lend themselves to managing relatively large-scale native prairie sites (parts of Missouri and Minnesota) or large pasture systems (Iowa, Missouri, Wisconsin). Some areas may contain unique complexes or concentrations of a variety of grassland habitats, and some Midwestern landscapes will require large-scale grassland restorations. Additionally, relatively widespread bird species such as eastern meadowlarks and vesper sparrows may benefit from management actions applied broadly across the region, whereas species with more localized distributions in the region such as sedge wrens and LeConte's and Henslow's sparrows will require much more targeted efforts.

States in the region should begin to identify large grassland areas and landscapes that contain significant grassland bird populations using existing land cover maps (including statewide GIS land use maps) and information such as bird data from the BBS and state Breeding Bird Atlas projects. To identify the type and location of habitats and landscapes important to the conservation of migratory grassland birds, some states may need to conduct additional surveys of bird distribution, abundance, and habitat preferences.

The question of how much habitat we need in the region to maintain or improve population sizes of grassland birds is unresolved. Also unsettled are questions regarding the spatial arrangement of conservation lands so as to maximize the benefits to grassland birds. Although there is little doubt that regional populations of many grassland bird species are declining, the underlying causes of these declines remain poorly known. For example, we do not know if population declines are due to the loss of breeding habitat or if they are the result of reproductive failure in existing habitats due to habitat degradation, mowing and other land-use changes or some combination of these factors. Additionally, potential problems on the wintering grounds and migratory routes must be considered as possible sources of population declines. Therefore, it is unclear if we should recommend to managers the creation of new grassland habitats for these species or if we should recommend management actions that seek to improve reproductive success in existing grassland areas. For most grassland birds we have too little information regarding their winter ecology to adequately assess the severity of threats on their winter grounds. For these reasons we recommend that conservation activities on the breeding grounds include both efforts directed toward enhancing habitat availability as well as actions directed toward improving grassland bird reproductive success in existing grassland areas in the region until more comprehensive data on other limiting factors become available and help focus management on specific problem areas.

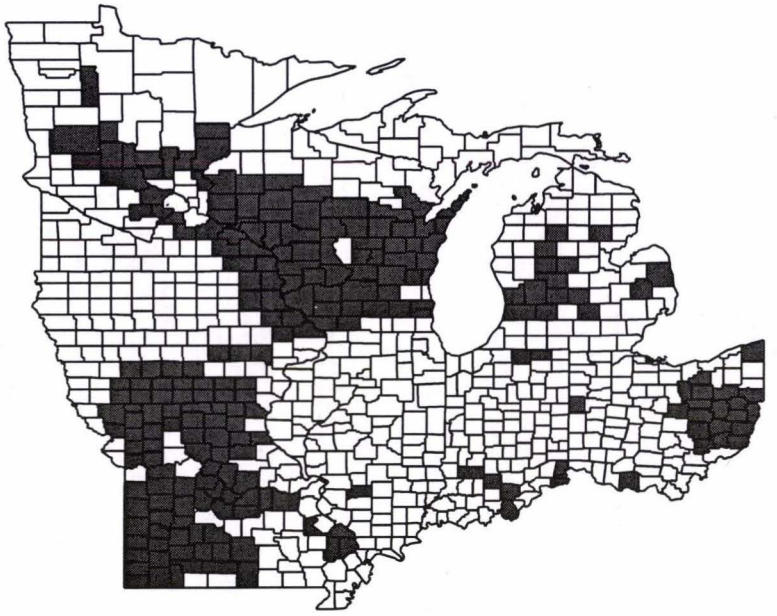

Fig 2a. Percent Hay (>6%)

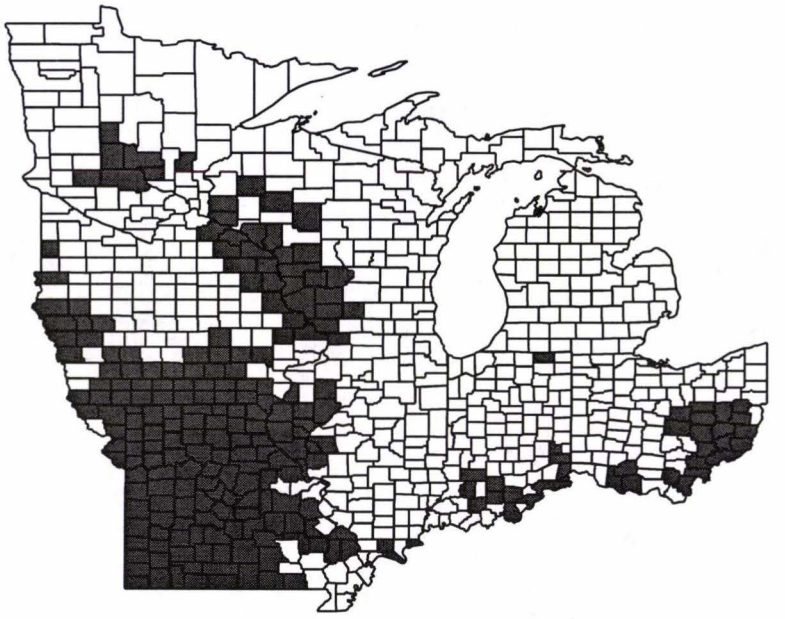

Fig. 2b. Percent Pasture (>8%)

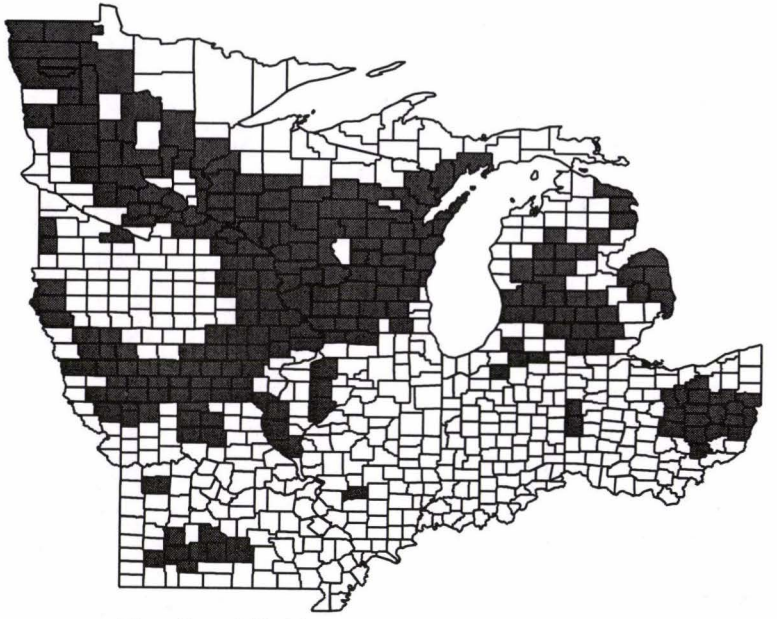

Fig. 2c. Alfalfa Acreage (>4,000 ha)

Figure 2.—*Distribution of counties in the Midwestern United States with significant amounts of grassland habitats, 1992. Data were compiled from the 1992 Census of Agriculture (U.S. Department of Commerce 1994).*

IMPACTS OF MAJOR MIDWESTERN LAND USE PRACTICES ON GRASSLAND BIRDS

Impacts of Hayfield and Pasture Acreage Loss on Midwestern Grassland Bird Populations

We used data from the North American Breeding Bird Survey (BBS) and the U.S. Census of Agriculture to examine the potential influence that regional changes in pasture and hayfield acreage may have had on Midwestern grassland birds over the last quarter century. The BBS consists of randomly located permanent survey routes established along secondary roads throughout the continental U.S. and southern Canada (Peterjohn and Sauer 1993). Survey routes are 39.4 km. long and consist of 50 stops at 0.8-km intervals. Each route is surveyed once annually, and all birds seen or heard within 0.4 km of each survey point during a 3-min census are recorded. In the Midwestern U.S. (Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, Ohio, and Wisconsin) there are 431 BBS routes that have been run since the late 1960s. Annual population indices for each species were extracted from the VU-BBS computer program (Hines *et al.* 1994) which were calculated using the residual method of Sauer and Geissler (1990).

We obtained data on changes in the acreage of Midwestern pastures and hayfields from the periodic U.S. Census of Agriculture conducted in 1964, 1969, 1974, 1978, 1982, 1987, and 1992 and published by the U.S. Department of Commerce. Estimates of pasture and hayfield acreage in intervening years were obtained by interpolating between known years assuming a constant rate of change within intervals. Comparisons of Illinois hayfield acreage estimates using this averaging method, with known annual acreage figures were highly correlated ($r = >0.90$, $n=30$), suggesting that these periodic data and this method may provide reliable annual acreage estimates for agricultural habitats.

Declines in the combined regional acreage of hayfields and pastures were significantly associated with declines in Midwestern grassland bird populations (fig. 3). The relationships between hayfield and pasture declines and concurrent grassland bird declines were strong; the correlation coefficients for the six species examined ranged from 0.76 to 0.94 (fig 3). More detailed analyses of these data suggested that population declines for these six Midwestern grassland bird species were more strongly associated with changes in pasture acreage than with changes in hayfield acreage (table 4).

Although we cannot discount the possible effects of other factors, our analyses (fig. 3) showed that population declines for several of the Midwest's most common grassland bird species are strongly correlated with regional declines in pasture and hayfield acreage, providing support to the idea that recent declines in these habitats have significantly influenced regional grassland bird populations.

Grassland Fragmentation

Several species of grassland birds avoid small grassland fragments and have populations in the Midwest that tend to be restricted to large grassland areas (Herkert 1994a). Migratory grassland bird species that appear to be influenced most by habitat fragmentation include northern harrier, short-eared owl, upland sandpiper, grasshopper sparrow, Henslow's sparrow, and bobolink, (Bollinger 1988, 1991; Herkert 1991c, 1992, 1994a; Herkert *et al.* 1993; Vickery *et al.* 1994). The estimates above are for minimal areas for species to occur at a site with some probability (generally >50 percent), not minimal areas required for self-sustaining populations. Both theoretical and empirical studies have shown that larger populations have a significantly greater probability of persistence, and that small populations are much more susceptible to local extinctions (e.g., Pimm *et al.* 1988, Berger 1990, Tracy and George 1992, Mangel and Tier 1994). Therefore, simply attracting small numbers of area-sensitive species to particular areas should not be the ultimate goal of grassland management. Rather, managers should strive toward providing habitat for large populations of area-sensitive species to increase the likelihood of long-term persistence for these populations. However, our knowledge of what constitutes a viable population size (and sources and sinks) for most all grassland species is very scarce. For this reason, areas that are much larger than a particular species' minimum area of occurrence will likely be required to adequately ensure the long-term persistence of area-sensitive grassland bird species. For some of the more area-sensitive grassland species

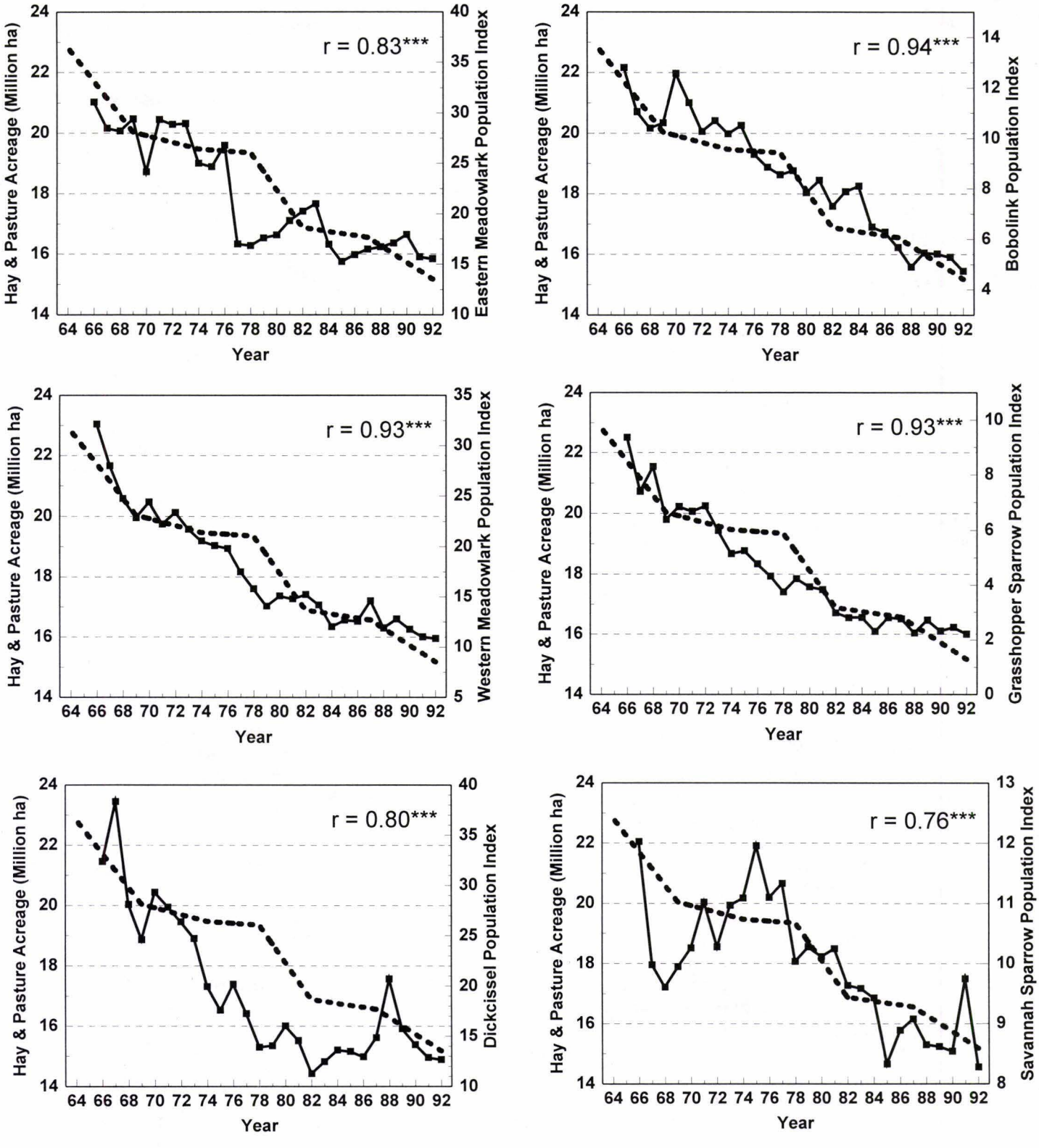

Figure 3.—*Correlations between combined Midwestern hay and pasture area and grassland bird population changes. Solid lines and squares represent grassland bird population indices. Pasture and hay area are shown with dashed lines. Grassland bird population changes were estimated from data from the Breeding Bird Survey. (*** = P<0.001.) See text for additional details.*

Table 4.—*Partial correlations between yearly estimates of regional grassland bird abundance and pasture and hayfield area. Pasture correlations are between annual grassland bird population indices and regional pasture area with the regional area of hayfields partialled out. Hayfields correlations are between bird indices and hayfield area with pasture area partialled out. Breeding bird indices were obtained from data from the Breeding Bird Survey. See text for additional details.*

Species	Pasture	Hayfields
Bobolink	0.94 ***	0.17
Dickcissel	0.85 ***	-0.26
Savannah sparrow	0.67 ***	0.35
Grasshopper sparrow	0.95 ***	-0.11
Eastern meadowlark	0.90 ***	-0.43 *
Western meadowlark	0.93 ***	-0.02

*** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$

these areas may be on the order of hundreds or possibly even thousands of hectares. For example, Temple (1992) determined in a modeling exercise that a population of sharp-tailed grouse, a non-migratory grassland species, would require 280 individuals and about 4,000 ha to be considered a viable population. Actual acreage required at a specific location will ultimately depend on the regional land use context of that particular area. In landscapes with minimal grassland cover in the surrounding landscape matrix, larger areas may be necessary, whereas small areas may suffice in landscapes with high proportions of grassland cover (see Sample and Mossman, in press).

Efforts to derive estimates of minimum habitat areas for grassland birds have suffered from a lack of consistent methods. Nevertheless, most estimates for minimum area requirements for grassland birds range from 10 to 100 ha, with some indication that a few of the larger, wide-ranging species may possibly require as much as 200 ha (Herkert 1994a, Vickery *et al.* 1994). Individual species' actual habitat area requirements, however, vary from location to location and may be greater for species near the edges of their range or where populations are very small and/or declining (O'Conner 1981, Smith and Smith 1992, Vickery *et al.* 1994).

Many grassland birds also appear to be influenced by regional landscape composition. For example, several grassland bird species may be more numerous in Midwestern landscapes that have a relatively high proportion of grassland cover such as pastures and idle grasslands (e.g., Rolley and Sample 1993). The distribution and abundance of these landscape features likely plays a strong role in determining regional grassland bird distribution and abundance patterns and also may influence species utilization of different size grassland fragments. For example, grassland birds may use smaller-sized fragments in landscapes with a higher proportion of grassland cover than in landscapes with minimal regional grassland cover. Such a relationship has been demonstrated for forest birds (Freemark and Collins 1992), but has yet to be shown for grassland bird species. In Wisconsin, however, grassland bird abundance has been shown to be influenced by factors in the landscape surrounding a site, such as mean patch size and amount of nearby idle grasslands (Sample and Mossman, in press).

Grassland fragmentation also influences grassland bird nest success. Nelson and Duebbert (1974) found that waterfowl nesting success was higher on large blocks of grassland habitat (32-48 ha) than on small blocks ($\leq$ 16 ha), due to high rates of nest predation in the smaller blocks. Johnson and Temple (1986, 1990) also found nest success to be lower on small prairie fragments (16-32 ha), than on large prairie fragments (130-486 ha) in Minnesota due to high rates of nest predation on their small prairie fragments. More recently, Greenwood *et al.* (1994) have suggested that bird nests in small blocks of grassland habitat are at higher risk of predation than are nests in large contiguous blocks of habitat that have not been fragmented. Davis (1994) also found nest success to be lower on his small site (22 ha) than it was on either of his two large (64 ha) sites. However, Davis' lower nest success was due to higher rates of nest parasitism on his smaller fragment. Nest predation rates were not significantly higher on his small site.

Grassland Management: Prescribed Burning, Grazing, and Mowing

Grasslands are disturbance-adapted systems. Historically, prairie fires occurred frequently and presently are recognized as one of the

most important types of disturbance essential for the maintenance of grassland ecosystems (Anderson 1970, Bragg 1982). Some form of regular disturbance appears to be important not just for grassland maintenance, but also for maintaining breeding bird diversity. Several studies have shown that grassland areas not subjected to periodic disturbances, such as fire or grazing, generally support both few grassland bird species and individuals (e.g., Kirsch and Kruse 1972, Skinner 1975, Whitmore 1981, Westemeier and Buhnerkempe 1983). In the absence of disturbance, grassland vegetative productivity declines and extensive invasion of woody plant species can occur (Risser *et al.* 1981, Hulbert 1986). Encroachment of woody vegetation onto grassland areas can occur rapidly, especially on small patches (Anderson 1970) and where residual forest cover (e.g., woodlots) or invading woody vegetation (e.g., from farmsteads, hedgerows, railroad lines, etc.) provide abundant seed sources of woody species.

Many grassland birds are negatively impacted by increases in woody vegetation or proximity to woody edges. Eastern meadowlarks, grasshopper sparrows, Henslow's sparrows, savannah sparrows, and Baird's sparrows all tend to avoid grassland areas that are being invaded by too much woody vegetation (e.g., Kahl *et al.* 1985, Arnold and Higgins 1986, Peterson 1983, Zimmerman 1988). Additionally, woody encroachment has been implicated as a factor in local grassland bird declines in studies in Iowa (Bernstein *et al.* 1990) and Massachusetts (Melvin 1994). Encroachment of grassland areas by woody vegetation has also been identified as a management and conservation problem in the southern United States (Hunter 1990, Telfer 1992), where many species of Midwestern migratory grassland bird species overwinter (American Ornithologists' Union 1983). In addition to the propensity for many grassland birds to avoid areas being invaded by woody vegetation, proximity to woody vegetation also has been shown to influence nest success. Grassland bird nests located close to woody vegetation experience higher rates of nest predation than do nests located far from woody vegetation (e.g., Johnson and Temple 1986, 1990; Burger *et al.* 1994; With 1994). Additionally, grassland bird nests located close to woody vegetation experienced higher rates of nest parasitism by brown headed cowbirds than nests located far from woody vegetation (Berger 1951, Best 1978, Johnson and Temple 1990).

As a result of the problems associated with encroachment of woody vegetation, managers now widely recognize the importance of regular management in maintaining grassland areas. However, grassland birds prefer a wide range of grass heights and densities; some species prefer short sparse vegetation whereas others prefer tall dense vegetation (e.g., Wiens 1969, Skinner *et al.* 1984, Sample 1989, Herkert 1991a, Herkert *et al.* 1993). As a result of these different habitat preferences, individual bird responses to various forms of grassland management are variable. Some species of grassland birds are more abundant in grasslands recently managed by fire, grazing or mowing, whereas others are more abundant in undisturbed areas.

Prescribed burning

Prescribed burning is the most commonly employed grassland management tool in the Midwest. Prescribed burning is an effective tool for suppressing woody encroachment, decreasing litter cover, and improving grass and forb production (reviewed by Ryan 1986).

Grassland birds whose abundance tends to be consistently lower immediately following a prescribed burn include sedge wren, Baird's sparrow, Henslow's sparrow, clay-colored sparrow, field sparrow, and dickcissel (Best 1979; Halvorsen and Anderson 1980; Huber and Steuter 1984; Zimmerman 1988, 1992; Pylypec 1991; Reinking and Hendricks 1993; Herkert 1994b, 1994c). Eastern meadowlarks, western meadowlarks, savannah sparrows and grasshopper sparrows show a variable, and possibly regionally dependent, short-term response to burning. Some studies have shown a positive short-term response to burning for these species while others have shown a negative short-term response (Tester and Marshall 1961, Huber and Steuter 1984, Pylypec 1991, Herkert 1994c). Grassland bird species that tend to be consistently more abundant on recently burned grasslands include upland sandpipers, killdeer, horned larks, chestnut-collard longspurs, vesper sparrows and lark sparrows (Renwald 1977, White 1983, Huber and Steuter 1984, Buhnerkempe and Westemeier 1988, Pylypec 1991, Zimmerman 1992). Some grassland bird species, such as bobolinks, have densities that are initially lowered by burning, but are usually higher in the 2 to 3 years following fire than they are in unburned areas (e.g., Cody 1985, Vickery 1993).

Burning also significantly effects grassland bird nest success. Nest success for a variety of ground nesting grassland birds is generally higher in recently burned grasslands than it is in unburned grasslands (e.g., Kirsch and Kruse 1972; Fritzell 1975; Johnson and Temple 1986, 1990). However, Toland (1986) found that nesting success for northern harriers in Missouri prairies was higher in unburned areas than in recently burned areas; and Reinking and Hendricks (1993) recently found nest success in burned tallgrass prairie areas to be variable from year-to-year and not consistently higher than unburned areas. Additionally, Best (1979) found that burning lowered cowbird parasitism rates on Illinois field sparrows. However, Johnson and Temple (1990) did not find a significant effect of burning on rates of cowbird parasitism in their study of breeding birds in Minnesota prairie fragments.

Few studies have attempted to assess the relative importance of other factors in determining grassland bird relative abundances within grassland areas subjected to prescribed burning. Herkert (1994c) found that habitat area had, in general, a stronger influence on grassland bird communities than did burn status for prairie fragments in Illinois. Precipitation patterns also may influence grassland bird response to burning, with individual species response to burning varying in relation to fluctuations in yearly precipitation (Zimmerman 1992). Factors such as the intensity and completeness of the burn and availability of adjacent refuge habitats also may influence grassland bird response to burning (Ryan 1986) but have received little attention to date. Additionally, timing of the burn (spring vs. fall) also may affect grassland bird response to fire. Higgins (1986) suggested that rates of nest success may be greater in grassland areas burned in the fall than in spring burned areas. Therefore, burning appears to be only one of a variety of factors that influence grassland bird distribution and abundance patterns within grasslands managed by prescribed fire. Further research is needed to more fully understand the relationship between burning and these other factors.

Grazing

Grazing is a versatile grassland management technique that can benefit many grassland bird species, particularly species which prefer short to medium vegetation heights (Skinner *et al.* 1984, Sample 1989). Grazing limits vegetation height, reduces litter accumulation, and also helps reduce woody encroachment (Sample 1989). In the Midwest, grazed grasslands tend to have more breeding grassland bird species and individuals than hayfields and even some idle grasslands (Skinner 1975, Sample 1989). Some species of migratory grassland bird species that are of high regional conservation concern, such as the grasshopper sparrow and bobolink, attain some of their highest regional densities in grazed grasslands (Sample 1989).

Grassland bird response to grazing varies regionally (Kantrud and Kologiski 1982). In the Midwest, shortgrass species such as the horned lark and killdeer favor heavily grazed areas, whereas upland sandpipers, grasshopper sparrows and eastern and western meadowlarks favor moderate grazing (Skinner 1975, Skinner *et al.* 1984). Many species tolerate light grazing, however moderate to heavy grazing greatly reduces and may even eliminate species such as northern harriers, short-eared owls, sedge wrens and Henslow's sparrows from grassland areas (Skinner 1975, Skinner *et al.* 1984, Kantrud and Higgins 1992, Lingle and Bedell 1989). Many factors can influence bird species response to grazing including soil types, soil moisture, plant species composition, weather, and stocking density and duration.

The effects of grazing on grassland bird nest success are not well known. Some studies have suggested that grazing during the breeding season may lower grassland bird nest success (e.g., Bowen and Kruse 1993, Kantrud and Higgins 1992, Kirsch and Higgins 1976). Other results show nest success varying regionally and according to grazing management practice. For example, Barker *et al.* (1990) found average nesting success for waterfowl in North Dakota was greater on all but one of five specialized pasture systems with low stocking rates than on idle grasslands; and with few exceptions it was greater than that believed necessary to sustain duck populations. Bientema and Muskens (1987) reported that trampling accounted for between 23 and 52 percent of total nest losses for four bird species in Dutch pastures and predicted that nesting success can be drastically reduced by high cattle densities. In Texas pastures with very low stocking rates, 84 to

100 percent of artificial bobwhite and turkey nests were lost, most to predation (Bariess *et al.* 1986). Using clay targets to simulate nests in Oklahoma pastures, Jensen *et al.* (1990) found that significant nest losses can result from trampling alone, especially with cattle densities above 10 head/hectare (similar to cattle densities in pastures in some parts of the Midwest).

Little work has been done on nest success in pastures in the Midwest. In Iowa warm season grass fields, pheasants and passerines hatched or fledged more young in idle fields than in fields grazed annually in July and August (George *et al.* 1979). Initial results from recent research in Wisconsin indicate that grassland bird nest success in continuously grazed pastures averaged 26 percent over 2 years (Stanley Temple, pers. comm.). Also in Wisconsin, an average of 73 percent of simulated pheasant nests were lost in pastures with a low-intensity rotation (9 head/ha) commonly used in dairy farming (Paine *et al.*, 1996). In general, pastures that are not managed to maximize variables such as forage, meat, or milk production are likely to have higher nest success (or at least fewer losses due to trampling), primarily due to lower stocking rates than when those variables are maximized. Further work on nest productivity in pastures under a variety of management regimes is needed.

Intensive rotational grazing (Undersander *et al.* 1991) is gaining popularity and acceptance in parts of the Midwest. Research in Wisconsin has shown that nest success in pastures that are under intensive rotational grazing is similar to or lower than that of conventionally grazed pastures (Paine *et al.*, in press; Stanley Temple, pers. comm.).

Mowing

Mowing, like grazing and burning, also can be used to lower vegetation height, reduce litter build up (if cuttings are harvested), and control woody vegetation (Sample 1989). Grassland bird response to mowing, when used as a management tool, is usually similar to their response to prescribed burning. Species whose abundance tends to be significantly reduced by burning, such as sedge wrens and Henslow's sparrows, also tend to be adversely affected by mowing (e.g., Herkert 1994c). The cutting and harvesting of hay during the nesting season, however, has serious consequences for breeding birds significantly reducing both breeding bird densities (Frawley and Best 1991) and reproductive success (Frawley 1989, Bollinger *et al.* 1990). Additionally, the harvesting of hay also may result in a loss of nutrients from the system and could result in altered grass and forb production (Ryan 1986). Overall, hayed grasslands support fewer bird species and individuals than grazed grasslands (Skinner *et al.* 1984). Several studies have shown that many grassland bird species abandon hayfields after their first cutting (Harrison 1974, Sample 1989, Frawley and Best 1991, Igl 1991). Some species, however, return to these hayfields after their first cutting and attempt to renest (Frawley 1989). Unfortunately, in many Midwestern hayfields there is generally not enough time between first and second cuttings for most species of grassland birds to raise young successfully (Frawley 1989; Herkert, in press). Cutting also has significant demographic consequences for grassland birds. In Iowa alfalfa fields, Frawley (1989) found that all above ground nests active at the time of cutting and 50 percent of all active ground nests were destroyed by hay mowing. In New York alfalfa hayfields, Bollinger *et al.* (1990) reported that mowing resulted in the failure of 94 percent of all bobolink nests active at the time of cutting. These losses to hay cutting can be a significant drain on grassland bird populations.

Although most attention of the effects of midseason cutting has been focused on agricultural hayfields, several other grassland habitats, such as public refuge lands and farm set-aside fields, are occasionally mowed during the nesting season (e.g., Strassman 1987, Hays and Farmer 1990) and also significantly reduce bird nesting success in these habitats. Several studies in the Midwest have reported an influx of birds into remaining uncut grasslands around the time that hayfields and roadsides are generally cut (e.g., Igl 1991, Bryan and Best 1991) pointing to the critical need for an increase in the acreage of uncut grasslands in the region. However, these secondary grassland areas also are occasionally cut creating a double jeopardy situation for breeding birds that may be attracted to them (Lou Best, pers. comm.).

If mowing is used as a management alternative, mowing must be timed to reduce the effects of cutting on breeding bird nest success. Ideally, mowing in areas that are being managed for grassland birds would not be conducted before 20 July. For each week that they are cut prior to this date, grassland bird potential reproductive success is reduced by approximately 5-10 percent (see fig. 2 in Frawley 1989:79). In agricultural hayfields, either delaying the first cutting of hayfields or an increase in the interval between successive cuttings would have obvious benefits for breeding birds. However, such recommendations are not realistic because they conflict with the objectives of farm operators to maximize their yields and to cut the forage crop when it has its peak value for livestock. This may be an example of an unreconcilable conflict, and we may have to accept the fact that some types of hayfields will always be ecological traps for breeding birds (Lou Best, pers. comm.).

Management implications: burning, grazing, and mowing

Due to the differing responses of grassland birds to the various forms of habitat management, managers are usually encouraged to provide a mosaic of burned and unburned, mowed and unmowed, or grazed and ungrazed grasslands to provide for the full range of grassland bird habitat preferences (e.g., Skinner *et al.* 1984, Ryan 1986, Renken and Dinsmore 1987, Herkert *et al.* 1993, Herkert 1994c). In such a plan, grassland areas are subdivided and managed in some type of rotational system in which some areas are burned, mowed, or grazed and other areas are left undisturbed. How frequently these subareas are disturbed is dependent on the management technique. Grasslands managed with prescribed burning or mowing should be burned or mowed often enough to prevent woody encroachment and possibly enhance nest success, but not so often as to negatively affect species, such as the Henslow's sparrow and sedge wren, which require areas that have not been recently disturbed. These vague recommendations do not translate easily into specific management prescriptions. This is due in part to the fact that there are still not enough data regarding individual species' responses to management to make robust predictions about the way that various species will respond to different management strategies under varying conditions. Moreover, most studies of the effects of burning on grassland birds have emphasized species short-term response to fire, and there is very little information available by which to predict how these short-term responses might be chained together to produce long-term patterns under different management scenarios. Based on the data at hand, however, it appears that burning or mowing 20-30 percent of a given grassland area every 3-5 years would sufficiently balance both the positive and negative impacts of burning or mowing in a way that would be most beneficial to migratory grassland birds. Just how large these subunits need to be is also not clear. However, it appears that subunits that are at least 30 ha or more in size stand the best chance of providing benefits to grassland bird species (Zimmerman 1988, Herkert 1994c). In bird conservation areas where grazing is the main management tool, some type of a rotational system is also usually recommended with a majority of areas being light to moderately grazed and with some closely grazed areas also provided in each season (Skinner *et al.* 1984).

With respect to woody encroachment there are currently few data regarding "threshold" levels of woody vegetation for grassland areas. Most grassland bird species tolerate some woody vegetation, and a few grassland bird species such as eastern meadowlarks, dickcissels, and vesper sparrows frequently use tall trees as song perches (Sample 1989). Research in Wisconsin has shown that grassland habitats with >4 percent total cover of woody vegetation >1 m begin to benefit woody or edge species over true grassland species (Sample 1989). In Missouri, grasshopper sparrows and Henslow's sparrows avoid areas with woody invasion >1 m tall (Kahl *et al.* 1985). Johnson and Odum (1956) reported grasshopper sparrows from early successional areas characterized by <10 percent coverage by shrubs. Based on these data, we recommend that managers strive to keep total cover of woody vegetation to <5 percent in areas being managed for grassland birds of open areas (see Sample and Mossman, in press). This recommendation also highlights the differing needs of grassland-associated bird species identified as of high regional concern in our prioritization scheme. Several high-ranking grassland-associated species, such as Bell's vireo and loggerhead shrike, are dependent on the presence of woody vegetation

in a grassland context for habitat suitability (Sample and Mossman 1994). Reducing woody vegetation too much on grassland areas will negatively influence these species. Therefore, there is also a need for the creation and maintenance of "shrub grassland" areas that focus on providing for the specific habitat requirements of these species in addition to providing grasslands areas that are open and free of woody vegetation for the "true" grassland birds. Providing habitat for shrub-prairie and open-grassland birds will require large areas if both habitats are to be included in the same area. An alternative is to manage specific areas as either open grassland or shrub prairie. In this situation, regional coordination would be needed to ensure that adequate amounts of both habitat types are available in the regional landscape in appropriate locations (see Ryan 1990).

Water-level Manipulation

Moist-soil habitats such as wet prairies, wet oldfields, and sedge meadows are affected not only by conventional grassland management activities but also by water level manipulations. For example, in Wisconsin ditching and draining sedge meadows lowers the local water table and can result (without other disturbances) in succession to shrub and tree-dominated communities, while flooding wet grasslands or sedge meadows to create impoundments or cranberry bogs can lower habitat quality for grassland birds (Mossman and Sample 1990). Thus, water-level manipulations must be carefully managed to maintain wet grassland and sedge communities.

LIMITING FACTORS

Temple (1988) identified three categories of factors that influence breeding bird populations and have the potential to influence long-term population trends; availability of breeding habitat, reproductive failure, and overwinter mortality. Loss and degradation of habitat are generally considered the most serious conservation problem facing species in North America (Noss and Murphy 1995). Many authors have implied that a decline in the acreage of native prairie and more recently agricultural grassland habitats has contributed to Midwestern grassland bird declines (e.g., Mayfield 1989, Sample 1989, Herkert 1991c, Warner 1994).

Our analysis of regional grassland bird population trends (fig. 3) suggests that loss of agricultural grassland habitats may be significantly influencing population declines for some species of Midwestern grassland birds. A few studies, however, have suggested that there is structurally suitable habitat that is unoccupied by some Midwestern grassland bird species (e.g., Brooks and Temple 1990, Basili and Temple 1995) suggesting that other factors besides habitat loss are also creating problems for migratory grassland birds in the Midwest.

Nest success for grassland birds is generally low, reports of nest success are frequently $\leq$ 30 percent (e.g., Roseberry and Klimstra 1970, Best 1978, Steigman 1990, Bryan and Best 1994, Camp and Best 1994). Several studies have suggested that nest success for grassland bird populations were so low that it was unlikely that adults were replacing themselves, and that these populations seemed dependent on immigration from other areas for population maintenance (e.g., Best 1978, Wray *et al.* 1982, Johnson and Temple 1986, Frawley 1989). Predation is generally considered the major cause of nest failure for grassland birds (Johnson and Temple 1986, Greer and Anderson 1989, Kantrud and Higgins 1992, Bowen and Kruse 1993, Bryan and Best 1994). However, in some Midwestern habitats mowing/haying is the primary source of nest losses (e.g., Frawley 1989). In pastures, trampling can also be a significant source of nest loss. Research in Wisconsin suggests that nest losses from trampling range from 25 percent (natural nests; Stanley Temple, pers. comm.) to 75 percent (simulated nests; Paine *et al.*, in press). Rates of nest parasitism by brown-headed cowbirds are variable among grassland birds, in some areas parasitism rates can be high (e.g., Hergenrader 1962, Hill 1976, Elliot 1978, Camp and Best 1994), but in other areas nest parasitism rates can be low (Robinson *et al.*, in press).

Limited winter habitat and/or winter resources have also been implicated in the declines of some grassland bird species (e.g., Fretwell 1986, Bucher and Nores 1988, Lymn and Temple 1991, Basili and Temple 1995). Fretwell (1972, 1986) suggested that dickcissel numbers, at least historically, and possibly still are winter-limited by the supply of seeds on their tropical winter areas. More recently, Basili and Temple (1995) have suggested that

efforts to control dickcissel's on their South American wintering grounds, where they are perceived as crop pests, may be behind recent population declines for this species. In the southeastern United States, the amount of suitable winter grassland habitat also has decreased substantially over the last 50 years due to conversion to croplands and pine plantations, and woody encroachment (Hunter 1990, Lymn and Temple 1991). This habitat loss also may be contributing to population declines since most declining Midwestern grassland birds winter primarily or partially in the southeastern Gulf Coast region (Herkert 1994a). Similar loss and conversion of grassland habitats are also occurring in major South American wintering areas used by Midwestern migratory grassland birds such as upland sandpipers and bobolinks (e.g., Bucher and Nores 1988, White 1988).

In summary, there is evidence that declines in the availability of grassland habitat, reproductive failure (due primarily to high rates of nest predation and occasionally nest parasitism), and problems on the North and South American winter grounds are all potentially influencing Midwestern grassland bird populations. Efforts to identify which factors are most important in limiting grassland bird populations are hampered by the limited data on aspects of their ecology. More data on grassland bird demographics are needed to identify and differentiate habitats that are sources from those that are population sinks, and a greater understanding of the winter ecology for most grassland species is also needed.

CONSERVATION STRATEGIES AND HABITAT MANAGEMENT GUIDELINES

We recommend that regional conservation efforts focus on all three of the potential limiting factors discussed above (habitat availability, reproductive failure, and winter ecology) until more information identifying specific problem areas for particular species becomes available. Most of our recommendations below focus on the breeding grounds although we recognize a serious need for more detailed information regarding the winter ecology for most of these species.

Our recommendations are in two broad groups. Our "conservation strategies" section includes recommendations that pertain to large-scale conservation and regional planning efforts. Often these are actions that transcend state boundaries but would be initially carried out by states or other local working groups, and at some point would need to be compiled into a comprehensive regional conservation plan. Our second group of "management guidelines" pertains to particular actions that land managers can carry out with the goal of benefiting local populations of grassland birds.

Conservation Strategies

- Stabilize the regional acreage of declining and undisturbed grassland habitats (i.e., pasture, old fields, native prairie, etc.), which may require some efforts to influence agricultural policies which significantly influences the amount of pasture, old field, and fallow areas on the landscape. At a minimum we recommend a "no net loss" principle for acreage of these major grassland habitats in the region. However, most likely the conservation of grassland birds in the region will require additional acreage of both native and non-native grasslands, especially some of the presently more restricted habitat types such as sedge meadows, wet-prairie, dry-prairie, shrub-prairie and open barrens.

- Maintain grasslands with a broad range of structure, plant species composition, field ages, moisture regimes, and topography in the regional landscape matrix. Ensure inclusion of habitats important for all species of local management concern.

- Increase conservation attention for the preservation of existing native grassland remnants. Wherever possible native remnants should form the core of restoration efforts targeted toward increasing the acreage of prairie and other native grasslands in the regional landscape. Incorporation of native remnant vegetation in habitat planning will promote the conservation of other components of native grassland biota (Henderson and Sample 1995).

- Initiate large-scale restorations (> 600 ha) in areas of the region that presently have limited grassland habitats available in the local landscape. Grassland bird use of these restorations should be closely monitored and used to guide future grassland restorations. A few large scale restorations are already underway in the region including

the 2,000-3,000 ha Walnut Creek National Wildlife Refuge (U.S. Fish and Wildlife Service) in Iowa and the 7,500 ha Midewin National Tallgrass Prairie (U.S. Forest Service) in Illinois.

- States should identify grassland areas and landscapes that possess significant grassland bird populations or have high grassland restoration potential. These landscapes should be large (>1,000 ha) and as treeless and open in character as possible (Henderson and Sample 1995).

- Initiate studies that relate grassland bird nest success to local habitat and landscape features so that regional "source" areas (or habitats) can be identified and so that features associated with low grassland bird reproductive success can be better identified. Habitat-specific productivity estimates are needed so that restoration and management efforts can be directed toward providing the most productive habitats for the species of concern.

- Support research addressing the winter ecology of migratory grassland birds that breed in the region.

- Maintain some managed upland grassland/shrub communities in appropriate parts of the region to meet the needs of bird species requiring a grassland/shrubland landscape matrix. Many of the region's open lands wildlife management areas have high potential for this and may, with minor modifications, be able to provide abundant habitat for these species requiring shrubland (S. K. Robinson, pers. comm.).

- Develop a regional assessment of the distribution, quantity, and quality of grassland habitat and the distribution, abundance, and habitat preference of grassland birds to determine where in the region it is most appropriate to manage for bird species and communities of management concern.

Management Guidelines

- Avoid fragmentation of existing grassland areas. The preservation and proper management of existing grassland areas, especially those presently used by area-sensitive, rare, or declining species is the most effective means of providing quality habitat for grassland birds (Herkert *et al.* 1993).

- Use some form of a rotational management system to provide a mosaic of burned and unburned, mowed and unmowed, or grazed and ungrazed grasslands to provide for the full range of grassland bird habitat preferences on large sites or among nearby small sites.

- When possible, locate managed grasslands adjacent to existing grassland habitat to increase overall size of grassland habitat blocks (Herkert *et al.* 1993, Sample and Mossman, in press).

- Eliminate mid-season cutting (prior to 20 July) of grasslands on public lands and agricultural set-aside lands.

- Aggressively control, and where necessary remove, woody encroachment in grassland areas being managed for open country grassland birds. Keep overall woody cover below 5 percent.

- Recognize the special features and value of your particular grassland areas. Avoid the temptation to provide habitat for all grassland birds in the region on all sites. Some sites may be best suited for management of low diversity grassland habitats (e.g., sedge meadows) that provide important habitat for specialist species that are rare in the regional avafauna (Sample and Mossman, in press).

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MANAGEMENT OF CENTRAL HARDWOOD LANDSCAPES FOR THE CONSERVATION OF MIGRATORY BIRDS

Frank R. Thompson, III[1], Scott K. Robinson[2], Donald R. Whitehead[3], and Jeffrey D. Brawn[2]

ABSTRACT.— We review major issues in the conservation of neotropical migratory landbirds (NTMB) in Central Hardwood landscapes, with special emphasis on Illinois, Missouri, Indiana, and Ohio. The Central Hardwood Region is dominated by oak forests, but some other forest types such as bottomland hardwoods and oak-pine forests also occur within the region. Historically fire and logging had a major effect on these forests. Of the 128 migratory landbirds that breed in the Central Hardwood States, 70 depend upon forest. Based on the Breeding Bird Survey, roughly equal numbers of species in each State are increasing and decreasing in abundance. Species of high management concern include those that depend upon mature forest and those that depend upon large-scale disturbances to create second-growth, shrub, and savanna habitats. Nest predation and cowbird parasitism levels vary greatly and are related to tract size and forest cover; the whole region may be characterized by "source-sink" population dynamics. The largest forest tracts in the region have the lowest levels of nest predation and brood parasitism and may produce the surplus of young that maintain populations in small tracts, which experience very high levels of parasitism and predation. Forest management practices affect both the abundance and nesting success of NTMB's. The effects of even-aged and uneven-aged silviculture on bird communities differ at the stand and landscape level. The impact of burning on forest birds is not well known, although it may be crucial to maintaining both oak-dominated forests and birds of open forests. Only a few large, heavily forested landscapes remain in the Central Hardwood Region. We believe that a prudent conservation strategy is to maintain a high percentage of forest cover in the few remaining heavily forested landscapes in the Midwest, to maintain the largest forest tracts in the region, and to take advantage of any opportunities to restore and recreate larger forest tracts in more fragmented landscapes. At a more local level land managers should provide appropriate habitats and landscape patterns by planning forest management activities to meet habitat goals, reducing internal edge and disturbances where appropriate, and planning comprehensively across ownerships.

INTRODUCTION

Neotropical migratory landbirds represent a large proportion of the bird community in Midwestern forests. They have recently been the focus of great conservation concern because of declines in some species and concern about the impacts of land-use practices and habitat fragmentation. We review the status and conservation of migrant birds in forested landscapes of the central Midwestern United States. We begin by reviewing the composition, distribution, and ecology of the forests in this region. We describe dominant land-use practices and forest management practices in the region. We then describe the forest bird communities in the region in terms of their abundance and population trends, and review species identified as species of high management concern by the Partners in Flight program. Next, we examine the impacts of land-use practices and disturbances on birds in this

[1] *North Central Forest Experiment Station, I-26 Agriculture Bldg., Columbia, MO 65211.*

[2] *Illinois Natural History Survey, 607 East Peabody Drive, Champaign, IL 61820.*

[3] *Department of Biology, Indiana University, Bloomington, IN 47405.*

region, including regional patterns in forest fragmentation and likely source-sink population dynamics. We conclude by synthesizing this information into a conservation strategy for migrant songbirds in the Central Hardwood region.

CENTRAL HARDWOOD REGION

Our focus is predominately on the Central Hardwood Region. This region has been defined as the general area included in the extent of oak-hickory forest type (Sander and Fischer 1989) and it largely corresponds to the Eastern Broadleaf Forest (Continental) Province and Prairie Parkland (Temperate) Province of the United States (McNab and Avers 1994). Because many of the statistics we review here are available by State as opposed to ecological boundaries and because of the midwestern focus of this symposium, we concentrated on the States of Illinois, Indiana, Missouri, and Ohio in this paper.

Forest Types

Oak-hickory forest dominates this area but gives way to mixed hardwoods in the east and oak-pine forest to the south. Bottomland hardwood forest also extends into the region (see Knutson *et al.* 1996). Oak-hickory forest covers approximately 50,166,000 ha or about one-third of all forest in the Eastern United States (Smith *et al.* 1994). The dominant tree species are white, black, scarlet, and northern red oak. Other overstory trees are southern red, post, blackjack, chinkapin, bur, and northern pin oak; bitternut, mockernut, pignut, and shagbark hickories; black gum; yellow poplar; red and sugar maples; white ash; elms; American beech; black walnut; and black cherry. The most common understory trees or shrubs are flowering dogwood, sassafras, redbud, serviceberries, eastern hophornbeam, American hornbeam, witch-hazel, blueberries, mountain-laurel, and beaked hazel (Braun 1950, Eyre 1980, Sander and Fischer 1989).

Oak-pine forests cover 12,932,000 ha in the Eastern United States. This forest type is very similar to the oak-hickory type except shortleaf, loblolly, pitch, and Virginia pine make up 25 to 50 percent of the forest. Mixed hardwoods or the maple-beech-birch forest type are found on moister, more productive sites primarily east of the Mississippi River. Within the Central Hardwood Region beech and birch are minor components. Principal species are yellow poplar, white oak, northern red oak, and sugar maple. Many other species are usually present including white ash, black oak, chestnut oak, red maple, black gum, basswood, buckeyes, black walnut, black cherry, and hickories. Common understory species include flowering dogwood, eastern redbud, serviceberries, sourwood, and sassafras (Braun 1950, Eyre 1980, Sander and Fischer 1989).

Forest History and Ecology

Recent reviews leave little doubt that disturbance has been central to the ecology and evolutionary history of forest ecosystems (Nuzzo 1986, Abrams 1992, Attiwill 1994) and the single most influential agent of periodic disturbance in forests worldwide has been fire (Attiwill 1994, Williams *et al.* 1994). In the Central Hardwood Region there is evidence of repeated burning of oak forests in the moister areas of the east and in the drier forest/prairie interface along the western boundary of the region. The history of uncontrolled burning ended less than a century ago in the Missouri Ozarks. Fires occurred there most frequently between 1785 and 1810 during an influx of Native Americans and early European settlers who practiced slash-and-burn agriculture. Fire frequency decreased with an exodus of Native Americans around 1815 (Guyette and Cutter 1991).

There is some debate about the extent that fire affected oak forests (Beilmann and Brenner 1951, Steyermark 1959) but there is evidence that fire-dependent communities, such as savannas, barrens, and glades, were more pervasive historically than they are now. Historical and paleoecological evidence indicates that these habitat types may have covered up to 13 million ha at the time of European settlement (Nuzzo 1986). In Missouri, oak savannas may have covered as much as one-third of the State (Nelson 1985). Although definitions for these habitats have proved elusive (see White 1994), it is clear that these open-canopy forests now are far less extensive. One estimate indicates that less than 20 percent of the presettlement acreage remains, but most of this has been degraded by over-grazing, fire suppression, and agriculture (Botts *et al.* 1994). Less than 1 percent of the former acreage remains in a high-quality state (Nuzzo 1986).

Logging also substantially affected the area through the 1800's as it followed human expansion. For instance, with the influx of settlers into the Ozarks from 1850 to 1874 came the railroad and a period of accelerated logging during the 1880's (Smith and Petit 1988). The lumber boom came to an end in the Ozarks in the early 1900's, and lumber companies sold much of the land sight unseen through the mail as farmland. Burning and open-range grazing of the forest occurred up until the 1930's in parts of the region. During the 1930's, much of this land became tax delinquent and was purchased by the US Forest Service. This land-use history resulted in widespread oak-dominated stands, many of which are not ecologically stable and will succeed to more tolerant hardwoods without disturbance (Parker and Weaver 1989, Johnson 1993).

Through most of the 20th century the amount of forest in the region declined due to clearing for agriculture and urban expansion. But there are now indications that this decline has leveled off and is reversing in some parts of the region (fig. 1). Recent forest inventories of the Central Hardwood States show 7.5- to 31-percent increases, per State, in the amount of forest land from the 1960's to 1980's (Raile and Leatherberry 1988, Smith and Golitz 1988, Brand and Walkowiak 1991, Hahn and Spencer 1991). The increase in forest land is primarily the result of farmland and pasture reverting to forest. In addition, Central Hardwood forests are becoming older and shifting to shade-tolerant species such as maple and beech (Raile and Leatherberry 1988, Smith and Golitz 1988).

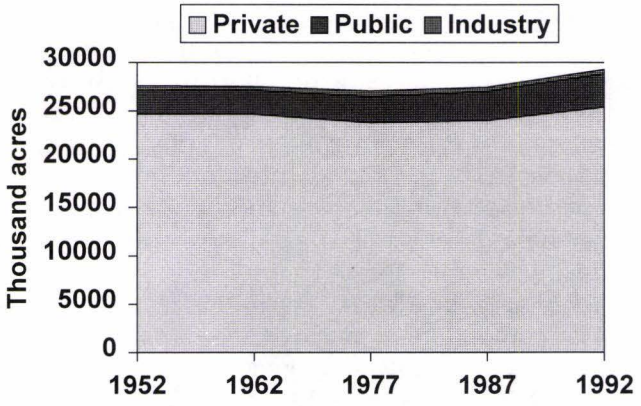


Figure 1.—*Trends in forest area in Missouri, Illinois, Indiana, and Ohio by type of ownership (Powell* et al. *1993).*

The dominance of oaks in Central Hardwood forests is largely the result of their adaptation to disturbance. Oaks in general are intolerant to moderately tolerant of shade. In most forests in the absence of major canopy disturbances, the amount of oak will decline and the numbers of more shade-tolerant species such as maples, blackgum, elms, beech, dogwoods, and redbud will increase. However, canopy disturbances allow light to reach the forest floor and allow young oaks, that have accumulated in the understory, to quickly gain canopy dominance. Recurrent fire promotes the accumulation of oak reproduction by eliminating or reducing the number of fire-sensitive understory competitors (young oaks survive by resprouting) and by reducing overstory density by killing trees with thin, fire-sensitive bark (Johnson 1993).

Not all oak-dominated ecosystems require fire or disturbance to persist. Many xeric oak forests appear to be relatively stable communities that show little evidence of succession to more shade-tolerant or mesophytic species. Probably the largest North American ecosystem of this type occurs in the Ozark Plateau in Missouri where oak reproduction may accumulate in the understory for 50 years or longer. Obtaining the accumulation of oak reproduction necessary for successful regeneration on highly productive sites requires recurrent disturbance. Historically, fire created the necessary conditions (Johnson 1993). Glades and oak savannas are even more dependent on fire than forests because an open canopy and herbaceous ground layer must be maintained by fire. Much evidence indicates that fire is essential to nutrient cycling, floristic composition, and structure of oak savannas (Botts *et al.* 1994).

The current composition of landscapes in the region differs greatly. The dominant land-use in the Region is agriculture and the amount of forest cover ranges from heavily forested to highly fragmented (fig. 2). We overlayed 1,000-ha (approximately) hexagons on a map of forest cover for Missouri, Illinois, Indiana, and Ohio. Only 6 percent of landscapes defined by these hexagons contained >80 percent forest cover and 63 percent had <20 percent forest cover (fig. 3).

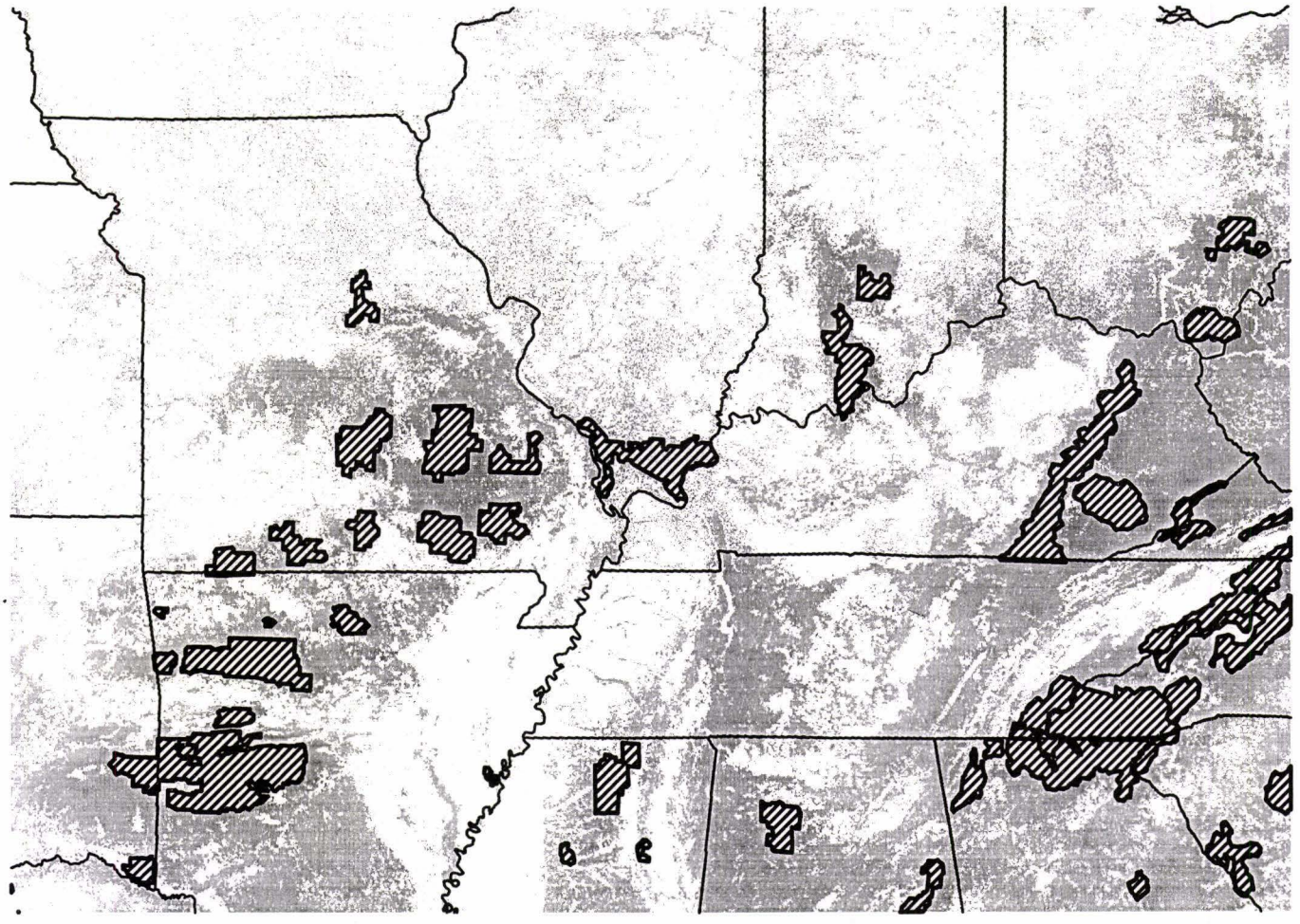


Figure 2.—*Distribution of forest cover in a portion of the Central Hardwood Region and the location of national forest boundaries.*

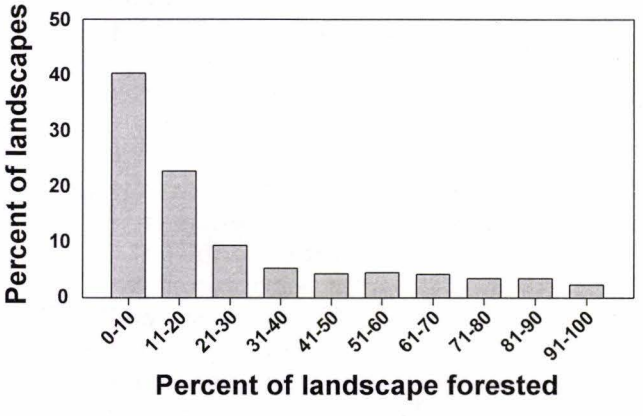


Figure 3.—*The percent forest cover in Central Hardwood landscapes.*

CENTRAL HARDWOOD BIRD COMMUNITIES

There are 128 long- and short-distance migrant landbirds that breed in the Central Hardwood States of Illinois, Indiana, Ohio, and Missouri (Appendix 1). Approximately 70 of these species breed in the forest. We report trends and abundances of many of these species from the Breeding Bird Survey for Illinois, Indiana, Ohio, and Missouri (Appendix 2). Caution, however, should be used when interpreting these data because the BBS is a roadside survey and does not always sample forest-interior birds adequately in these States. For instance, relatively common forest species such as the ovenbird are not detected in several States. Six to 14 forest species in each of these States show long-term population declines, and 5 to 16 species in each State show long-term population increases (Appendix 2).

We identified forest-breeding species of high management concern from the Partners In Flight Database for Illinois, Indiana, Ohio, and Missouri (Carter and Barker 1993) (table 1). Species in the database are scored on 7 criteria reflecting abundance, distribution, population trend, and population threats (Hunter *et al.* 1993). Scores range from 7 to 35; we report species with scores ≥21 as possible management priorities within this region (a score of 21 usually is about the 75th percentile). Species on these lists consistently use a variety of forest habitats. Some of the highest scored species (Bachman's, cerulean, prothonotary, and Swainson's warblers) breed in lowland forest. Golden-winged, blue-winged, chestnut-sided, and prairie warblers breed in young forest or shrub habitats. Worm-eating, black-throated blue, blackburnian, hooded, Northern parula, and Canada warblers; Acadian and great-crested flycatchers; wood thrushes; and scarlet tanagers breed in mid- to late-successional forest.

We synthesized information on abundance of Midwestern NTMBs in forest habitats ranging from glades to mature forest (fig. 4). This list includes common species as well as species identified as a high management concern. We reviewed research and monitoring studies in this region (Annand and Thompson 1997 ; Chambers 1994; S. Robinson, unpubl. data; J. Brawn, unpubl. data; D. Whitehead, unpubl. data) to make our own qualitative assessment of the abundances of these species in a variety of Central Hardwood forest habitats during the breeding season. Importantly, these abundances are only one indication of habitat quality. The quality of breeding habitat ultimately should be based on reproductive success; however, adequate information on the reproductive success of most species across a range of habitats is not available. Considerable differences in abundances of birds across this region complicate interpretations of habitat use, so land managers also must account for geographic patterns in abundance. Habitat relationships of NTMBs within the region varied greatly (fig. 4). Habitat relationships are discussed in the following sections in relation to management practices.

IMPORTANCE OF REGIONAL AND LANDSCAPE PATTERNS

Population size and viability are determined by interactions between local habitat factors and regional or landscape features. An individual may select a nest site based on micro-habitat, and the density of a species might be related to vegetation structure in a habitat patch (Martin 1992). Densities, however, also may be affected by biogeographic patterns in a species' distribution (Thompson *et al.*, in press), and reproductive success may be affected by landscape composition and structure (Robinson *et al.* 1995). Conservation plans, therefore, must address multiple spatial scales. In this section we discuss how regional- and landscape-scale patterns in land use affect forest birds. In the following section we address the effects of forest management on local and landscape factors that affect forest bird populations.

Landscape composition and pattern significantly affect the reproductive success and status of forest bird populations in the Central Hardwood Region. Productivity, source-sink status of populations, or levels of nest depredation or brood parasitism are related to landscape patterns in forest cover (Donovan *et al.* 1995a; Robinson *et al.* 1995; Thompson *et al.*, in press). These studies characterized landscape patterns with measurements such as percent forest cover and mean patch size, in areas defined by 1- to 10-km radii.

Strong regional patterns in the productivity of NTMB's in Central Hardwood forests occur because of the great variation in amount of forest cover (Robinson *et al.* 1995). Populations of forest birds in the Central Hardwood Region are likely structured as population sources and sinks (see Donovan *et al.* 1996). Reproductive rates of some forest birds are so low in some fragmented landscapes that these populations are likely "sinks" that cannot sustain themselves without immigration from more productive "source" populations (Donovan *et al.* 1995a, Robinson *et al.* 1995, Brawn and Robinson 1996). Reproductive rates appear high enough in heavily forested landscapes in the Missouri Ozarks, south-central Indiana, and northern Wisconsin for these areas to act as population sources (Donovan *et al.* 1995a, Robinson *et al.* 1995). Other heavily forested landscapes that potentially could act as sources for species breeding in Central Hardwood forests include portions of Tennessee, eastern Kentucky, and southern Ohio (fig. 2). The small number of large, heavily forested landscapes in the region may be sustaining populations throughout the

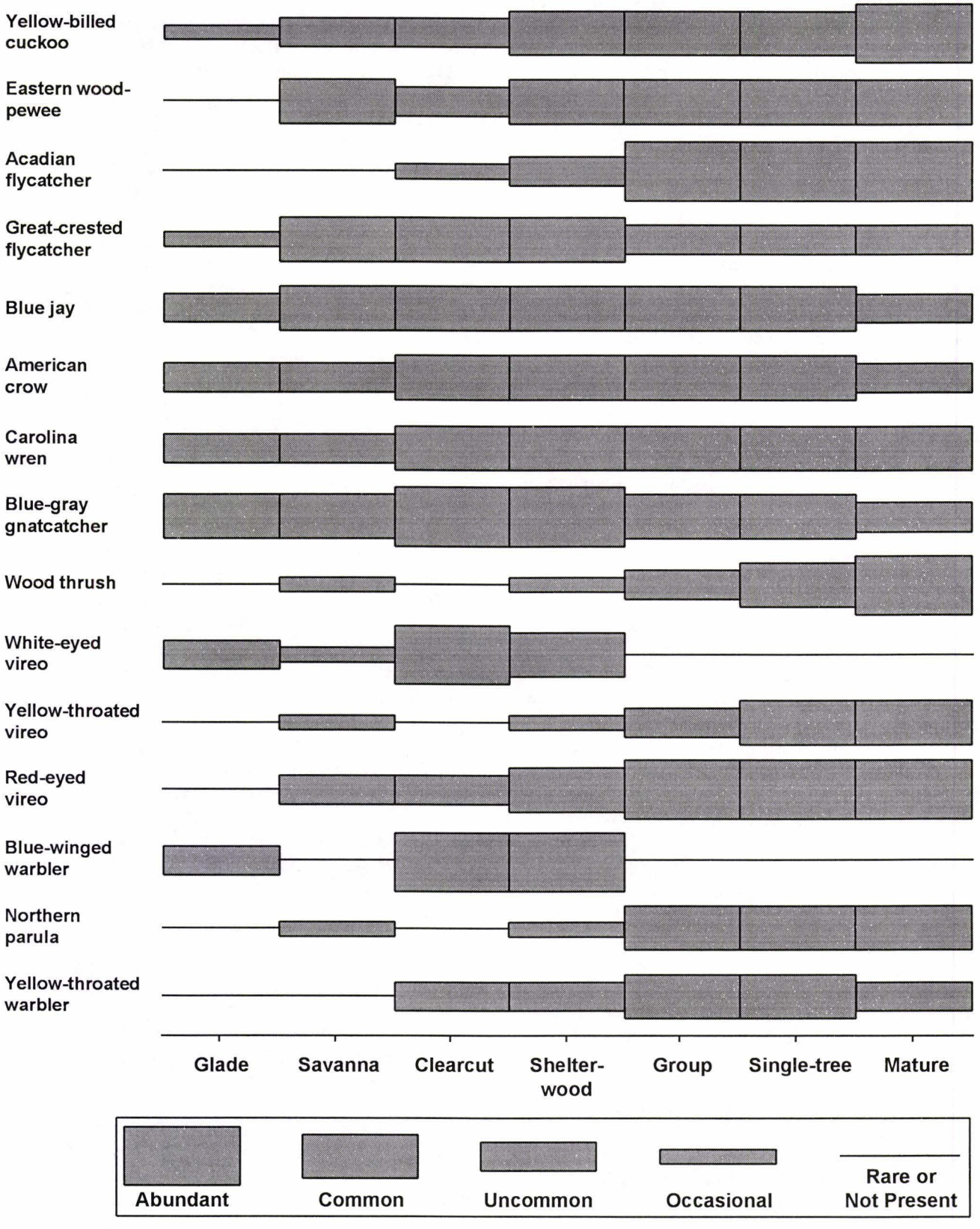


Figure 4.—*Habitat relationships of some breeding neotropical migratory birds in Central Hardwood habitats. Glades and savannas are managed by controlled burning; clearcut, shelterwood, group selection, and single-tree selection refer to forest stands treated within the last 10 years by these regeneration methods; and mature refers to 60- to 100-year-old even-aged forest.*

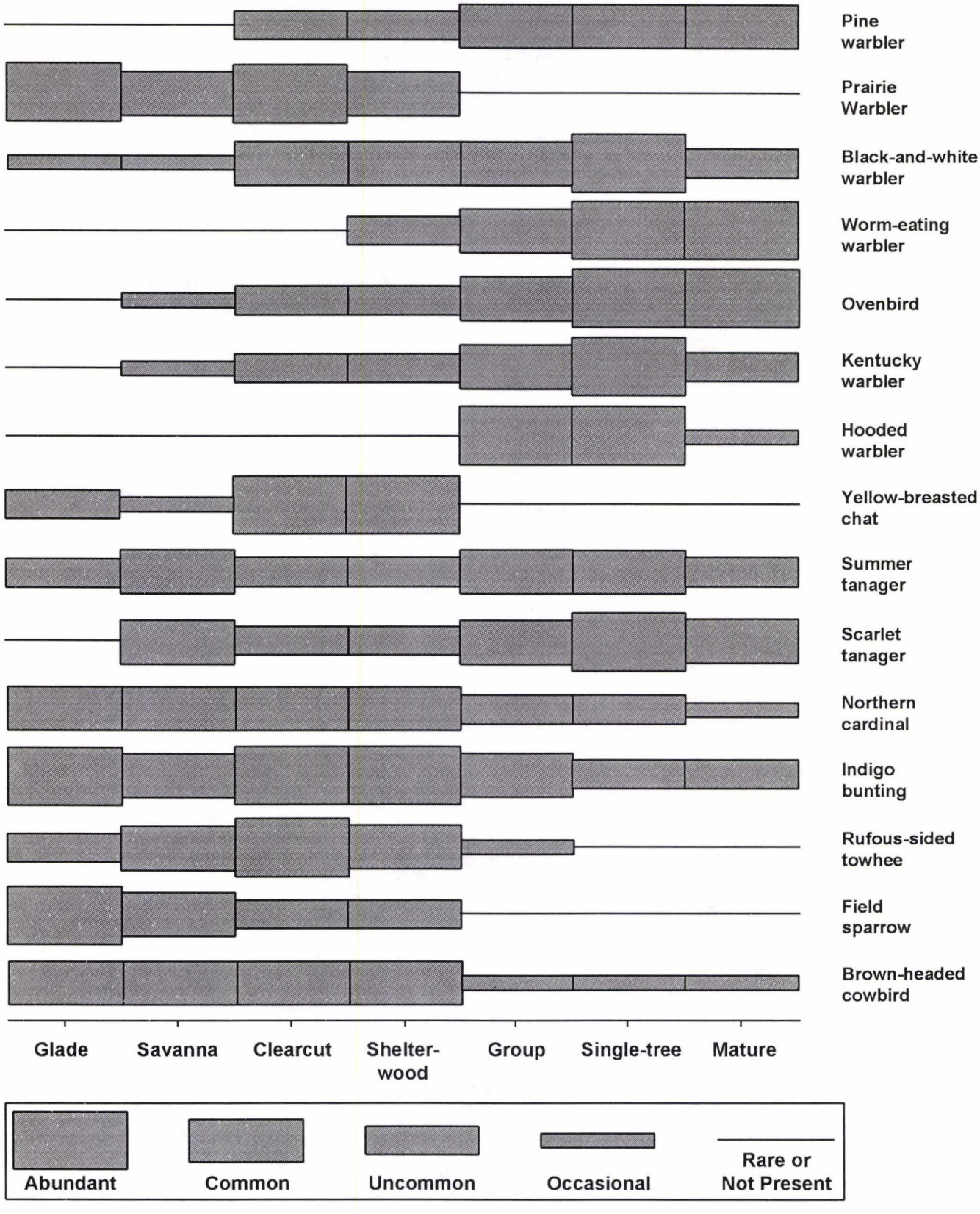

Pine warbler
Prairie Warbler
Black-and-white warbler
Worm-eating warbler
Ovenbird
Kentucky warbler
Hooded warbler
Yellow-breasted chat
Summer tanager
Scarlet tanager
Northern cardinal
Indigo bunting
Rufous-sided towhee
Field sparrow
Brown-headed cowbird
Glade
Savanna
Clearcut
Shelter-wood
Group
Single-tree
Mature
Abundant
Common
Uncommon
Occasional
Rare or Not Present

Table 1.—*Species of high management concern in 4 Midwestern States. Numbers are management concern scores generated by the Partners in Flight program in cooperation with the Colorado Bird Observatory and State conservation agenecies and Partners in Flight working groups. Scores can range from 7 to 35; species with scores >21 are presented as possible conservation priorities in this region.*

Illinois		Indiana		Missouri		Ohio	
25	Cerulean warbler	26	Cerulean warbler	29	Bachman's warbler	27	Cerulean warbler
25	Swainson's warbler	25	Golden-winged warbler	26	Cerulean warbler	26	Golden-winged warbler
24	Black-billed cuckoo	23	Wood thrush	25	Swainson's warbler	25	Blue-winged warbler
24	Golden-winged warbler	23	Blue-winged warbler	24	Prothonotary warbler	24	Chestnut-sided warbler
24	Chestnut-sided warbler	23	Chestnut-sided warbler	24	Worm-eating warbler	23	Great-crested flycatcher
23	Acadian flycatcher	23	Prothonotary warbler	23	Black-billed cuckoo	23	Prothonotary warbler
23	Great crested flycatcher	23	Worm-eating warbler	23	Wood thrush	23	Worm-eating warbler
23	Sedge wren	23	Louisiana waterthrush	23	Chestnut-sided warbler	23	Louisiana waterthrush
23	Wood thrush	22	Black-billed cuckoo	22	Yellow-billed cuckoo	22	Black-billed cuckoo
23	Blue-winged warbler	22	Yellow-billed cuckoo	22	Yellow-throated vireo	22	Yellow-billed cuckoo
23	Prothonotary warbler	22	Acadian flycatcher	22	Prairie warbler	22	Yellow-throated vireo
23	Worm-eating warbler	22	Hooded warbler	22	Louisiana waterthrush	22	Black-throated blue warbler
23	Louisiana waterthrush	21	Great crested flycatcher	22	Hooded warbler	22	Blackburnian warbler
22	Yellow-billed cuckoo	21	Canada warbler	21	Acadian flycatcher	21	Acadian flycatcher
22	Hooded warbler			21	Great crested flycatcher	21	Wood thrush
21	Canada warbler			21	Blue-winged warbler	21	Prairie warbler
21	Scarlet tanager			21	Northern parula	21	Hooded warbler
						21	Canada warbler
						21	Scarlet tanager

region. While observed variation in reproductive rates and population modeling corroborate source-sink population structure, there is no direct evidence of dispersal from sources to sinks.

Much of the forest habitat in the Central Hardwood Region could be sink habitat, and even though populations there essentially are nonviable and dependent on immigrants, they still may be important. A large portion of a species' global population may reside in population sinks at any one time. Sink populations may be important holding areas for surplus breeders (Howe *et al.* 1991). Individuals occupying sink habitat are involved in many ecological processes (Marquis and Whelan 1994), and these interactions may be important despite individual reproductive success.

Such large-scale source-sink population dynamics also could have major implications for interpreting census data such as those of the BBS. A lack of long-term declines in midwestern forest birds may not be an indicator of healthy populations (Brawn and Robinson 1996). Small, isolated woodlots in the Midwest often show no consistent long-term declines even though reproductive success may be far below the levels necessary to maintain local populations (Robinson 1992, Brawn and Robinson 1996). Interpreting results of population trend analyses therefore requires additional data on nesting success to be of most use in making management decisions.

The primary hypothesis for the relationship between the reproductive success of forest breeding birds and landscape composition is that predator and cowbird numbers are greater in landscapes where the forest is fragmented by human activities such as agriculture and development. It appears that any human activities that create cowbird and predator feeding habitat within an otherwise forested landscape will reduce NTMB nesting success. Even a single cowbird feeding area in forested landscapes can reduce nesting success over large areas because cowbirds can commute up to 7 km between feeding and breeding sites (Rothstein *et al.* 1984, Thompson 1994). In practice, however, mostly forested landscapes where there are few feeding opportunities for cowbirds have low numbers of cowbirds and parasitism levels (Thompson *et al.*, in press). Our working hypothesis is that cowbirds are limited by feeding habitat in mostly forested landscapes and by nesting habitat in mostly agricultural landscapes (Robinson *et al.* 1995; Thompson *et al.*, in press). In the agricultural landscapes of Illinois, cowbirds appear to saturate the available landscape and their abundance is determined by host abundance rather than by proximity to edge or pastures (Robinson and Wilcove 1994; Thompson *et al.*, in press).

Because nest predators are so diverse, regional and landscape patterns of nest predation generally are less pronounced than those for cowbird parasitism (Robinson *et al.* 1995). Predators such as raccoons and opossums that forage extensively in agricultural habitats may be a much greater problem in agricultural landscapes than in forested landscapes, as are some corvids (reviewed in Paton 1994, Marini *et al.* 1995). Some snakes also appear to be most abundant near agricultural openings (Durner and Gates 1993). Marini *et al.* (1995) and Heske (1995), however, found no differences in the abundance of mammalian nest predators between agricultural edges and forest interiors in southern Illinois. It is therefore possible that some landscapes may be saturated with nest predators just as they are with cowbirds. Nevertheless, reducing agricultural, suburban, and urban inholdings in forested landscapes likely would also reduce certain kinds of nest predation.

Edge Effects

Although regional effects of forest fragmentation and landscape composition on NTMB nesting success have been well-defined, there is also evidence that local features affect songbird nesting success (reviewed in Robinson 1996). Sometimes edges are associated with higher predation and parasitism levels (Brittingham and Temple 1983, Temple and Cary 1988). Some species in some regions experience variation in local reproductive success at least as great as those seen among landscapes across the region (Robinson *et al.* 1995; Trine *et al.*, in press). One hypothesis is that edge effects are responsible for variation in reproductive success among landscapes because more fragmented landscapes have more edge. While studies have demonstrated local edge-effects, the consequences of edge effects to NTMB populations are largely unknown and depend on whether predators and cowbirds respond numerically or functionally to edges,

and whether NTMB's are attracted to or avoid edges. If both predators or cowbirds and NTMB's are attracted to these edges, they could function as ecological traps (sensu Gates and Gysel 1978).

Edges between forest habitats and non-forest habitats (such as agricultural, suburban, grassland habitats) may affect species abundances' as well as reproductive success (we discuss edge effects from forest management practices in the following section). Some mature forest species such as ovenbirds and red-eyed vireos are less common near agricultural edges than the forest interior, and some early successional species such as blue jays and crows are more common near agricultural edges (Frazer 1992).

Research in southern Indiana compared reproductive success between interior sites (heavily forested sites >5 km from agricultural habitats) to exterior edge sites (forest adjacent to agricultural habitats) (D. Whitehead, unpubl. data). Overall nest survival rates tended to be lower near agricultural edges for 6 of the 7 species monitored. Differences for one species (Acadian flycatcher) approached significance (p=0.075), and across all species nest success was significantly lower near agricultural edges. Patterns of nest parasitism were similar; parasitism tended to be higher near edges for 6 of the 7 species. Differences were significant for 2 species (red-eyed vireo and worm-eating warbler), and across all species parasitism was significantly higher near exterior edges (fig. 5).

FOREST MANAGEMENT PRACTICES

Central Hardwood forests are disturbance-adapted ecosystems. Forest management activities, including timber harvest, wildlife management, and fire suppression or prescription, have to some degree replaced natural disturbance patterns. For nongame species, such as migratory birds, a predominant view has been that disturbance is not beneficial (e.g., see Morrison *et al.* 1992). Notwithstanding the clearly adverse effects of habitat loss and fragmentation caused by urbanization or agriculture (see Robinson *et al.* 1995), disturbance is often seen as "unnatural" and negative, especially for those species requiring old-growth. The "non-disturbance" view of habitat management for nongame birds has some merit for those species requiring large, old-growth forest stands, but we believe that disturbance plays a fundamental role in managing forest habitat and molding the structure of landscapes. For instance, small disturbances on the scale of a few trees create hooded warbler habitat, whereas large disturbances (>3 ha) may be required to create prairie warbler habitat.

In this section, we focus on a few dominant forest management practices in the Central Hardwood Region. We review the effects of silvicultural practices on NTMB's, both mitigating their negative effects and in using them to create NTMB habitat. We also present information on the role of fire in NTMB conservation. Fire increasingly is being used to maintain or restore fire-dependent ecosystems.

Impacts of Silviculture

Perhaps the most important management issue in the Central Hardwood Region is determining the best strategy for harvesting timber while conserving NTMB's and other ecosystem attributes (Kurzejeski *et al.* 1994). Central hardwood forests are managed by even-aged and uneven-aged silvicultural systems. These two systems have some fundamental differences in their effects on forest habitats and wildlife (see Thompson *et al.* 1995). In addition to these silvicultural practices, high grading, a practice common on private lands, may range from nearly complete removal of the existing stand (resembling clearcutting) to removal of a few marketable trees (resembling selection methods). The objective of high grading is to harvest marketable trees and not necessarily to regenerate a forest stand that will provide for future commodity or non-commodity values. Forest management generally is implemented at the stand level, resulting in a habitat patch. At this scale we discuss the types of habitats created and the dominant species that breed in them. We also evaluate practices at the landscape scale to assess cumulative impacts (both spatial and temporal) and impacts resulting from the spatial arrangement of habitats.

Even-aged Methods

These regeneration methods harvest an existing stand and result in a young even-aged forest. Throughout most of the Central Hardwood Region rotation ages are 80 to 100 years, although rotations sometimes are extended to

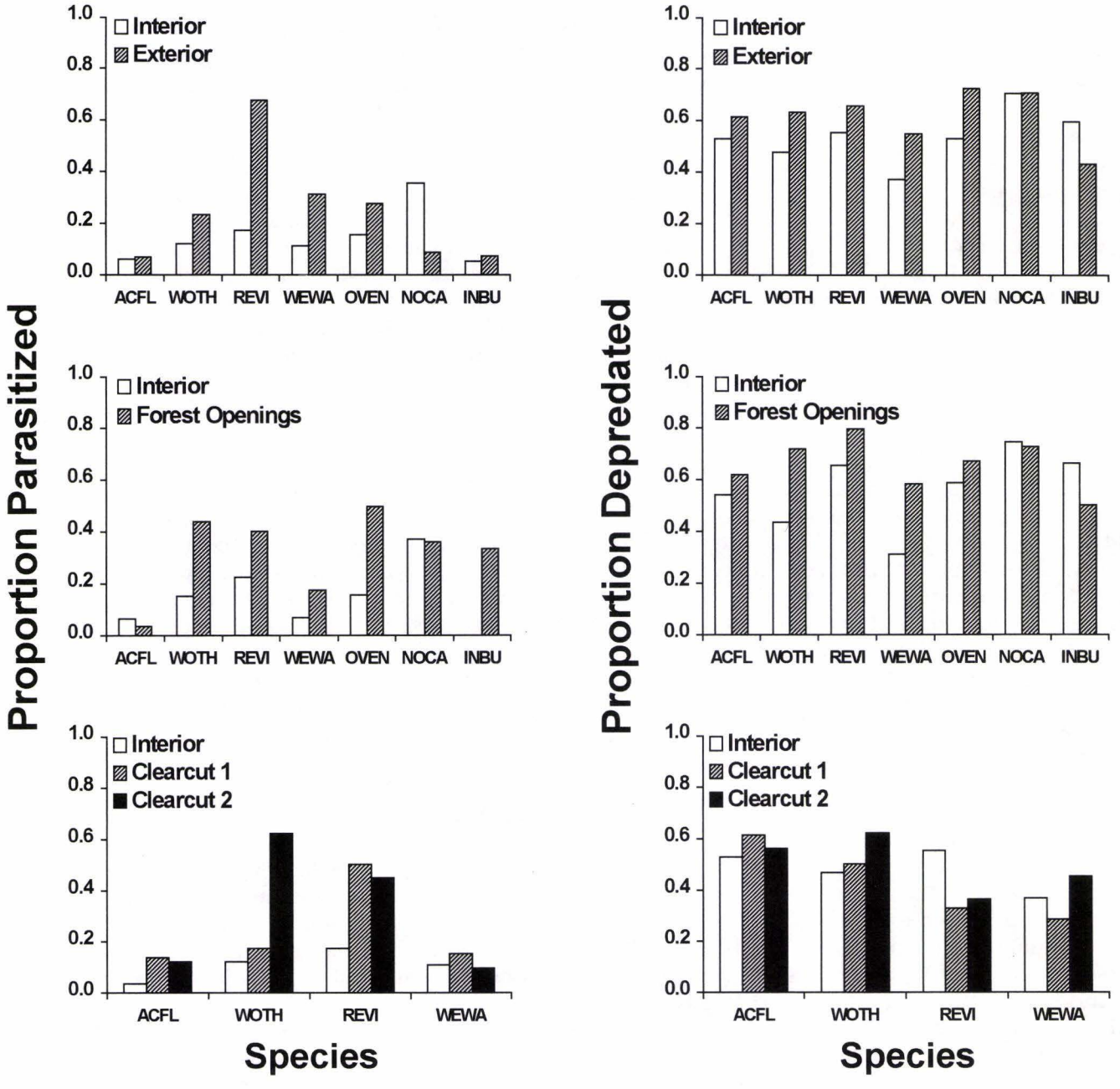


Figure 5.—*Differences in reproductive success of some forest songbirds in forest interior and in forest adjacent to different types of edge in southern Indiana (D. Whitehead, unpubl. data).*

provide stands with some old-growth characteristics. Even-aged regeneration methods differ primarily in the number of trees retained from the previous stand. Under the clearcut method, all trees are removed except for designated den trees or snags. Under the shelter-wood method, up to 60 percent of the canopy may be retained to provide shelter for developing tree reproduction. At some point early in the rotation the shelter generally is removed, but in some instances it may be retained to meet scenic quality, wildlife, or other objectives.

Regenerating stands

The first year after cutting, these stands have abundant herbaceous ground cover, but they quickly become dominated by tree regeneration from sprouting and advance regeneration, resulting in as many as 25,000 stems/ha (Gingrich 1971). The first year after clearcutting there is usually a drastic reduction in total bird numbers and a nearly complete turnover in species. American goldfinches and field sparrows often prefer cuts at this age because of the abundant annual and perennial vegetation. As tree regeneration begins, numerous other species nest in this habitat. NTMB species of high management concern that breed in these habitats include golden-winged, blue-winged, hooded, chestnut-sided, and prairie warblers. Other birds breeding in these habitats are yellow-breasted chats, indigo buntings, Kentucky warblers, common yellowthroats, white-eyed vireos, gray catbirds, rufous-sided towhees and northern cardinals (Gremel 1989, Frazer 1992, Thompson *et al.* 1992, Annand and Thompson 1997). In shelterwood and seed tree cuts, and clearcuts with residual live trees and snags, some mature-forest, canopy-dwelling species may continue to use the stand. Species richness and total bird density are high during this stage because of the mixture of foraging and nesting substrates (bare ground, grasses and herbs, shrubs and young trees, mature-residual live and dead trees).

Some have speculated that residual trees left in regeneration cuts provide perches for cowbirds and nest predators that might enable them to find nests within the openings. However, there is no direct evidence supporting this hypothesis. Residual trees do have demonstrated value for cavity-nesting wildlife, including NTMB's such as the great-crested flycatcher (Dickson *et al.* 1983).

Sapling stands

Between 10 and 20 years after harvest, the stand is dominated by tree saplings with a closed canopy. At age 20 the number of stems has been reduced to 3,400 to 6,200/ha and the larger trees on good sites have reached 18 cm d.b.h. (Gingrich 1971). Many birds typical of regeneration do not breed here or persist in these stands at lower densities. Black-and-white, worm-eating, and Kentucky warblers seem to prefer the high stem densities and closed canopies this age class provides. Ovenbirds, wood thrushes, and red-eyed vireos may begin using stands at this age (Gremel 1988, Frazer 1992, Thompson *et al.* 1992).

Poletimber stands

From age 20 to 60 years, 90 percent of the trees will die due to competition. The canopy remains closed and there is little understory development. As a result common species tend to be canopy nesters such as red-eyed vireos, scarlet tanagers, eastern wood-pewees, and wood thrushes, or ground nesters such as ovenbirds and black-and-white warblers. Total densities of NTMB's may be lower than in mature forest or regenerating stands (Gremel 1989, Thompson *et al.* 1992)

Mature stands

The structure of mature forests differs widely through the region. Depending on soils, geology, climate, and geography, mature stands may have sparse to dense groundcover and understory. Death and decay of large trees results in cavities, snags, and tree-fall gaps. After 80 years, dominant trees will range from 30 to 46 cm d.b.h. If left undisturbed, these stands will slowly become uneven-aged as they age and individual trees die. However, because of the widespread logging, burning, and grazing in this region in the late 1800's and early 1900's, much of the mature forest here is even-aged ranging from 60 to 100 years old. The most abundant NTMB in mature forests throughout the region is the red-eyed vireo. Other abundant or common species throughout the region are the eastern wood-pewee, wood

thrush, Acadian flycatcher, blue-gray gnatcatcher, ovenbird, worm-eating warbler, scarlet and summer tanager, and blue jay. In oak-pine stands pine warblers and yellow-throated warblers are common.

Community trends and succession

Avian density and diversity generally increase with plant succession in the Southeast US (Johnston and Odum 1956, Karr 1971, Shugart and James 1973, Shugart *et al.* 1975). However, breeding bird densities in regenerating oak hickory or pine hardwood stands usually are similar to or much greater than those in mature stands, and densities are often lowest in mid-successional sapling- and pole-sized stands (Conner and Adkisson 1975, Conner *et al.* 1979, Dixon and Selquist 1979, Probst 1979, Horn 1984, Yahner 1986, Gremel 1989, Thompson and Fritzell 1990, Frazer 1992). Species richness and diversity generally increase with stand age but sometimes also show an early peak in regenerating stands (Conner and Adkisson 1975, Conner *et al.* 1979, Dixon and Segelquist 1979, Probst 1979, Horn 1984, Yahner 1986, Thompson and Fritzell 1990).

Landscape composition and pattern

Even-aged management creates a mosaic of different-aged forest stands in the landscape. Assuming timber harvest is regulated to provide sustained yield over time, rotation age will determine the amount of forest in any given age class. For instance, an oak-hickory forest managed by regulated clearcutting on an 80-year rotation would be comprised of approximately 10 percent 1- to 8-year-old forest, 10 percent 9- to 16-year-old forest, etc. Forests managed by even-aged management could have more or less early successional forest than natural landscapes depending on rotation age and frequency of natural disturbances. Managed Central Hardwood forests, however, generally have more early successional forest and bird species than forest reserved from harvest. In the Missouri Ozarks, for example, the amount of young forest and density of early-successional NTMB's was greater, and the density of forest-interior NTMB's slightly lower, in landscapes managed by clearcutting than those with no recent timber harvest (Thompson *et al.* 1992).

Edge effects

Edges between regenerating stands and mature stands may affect species abundance and diversity. For instance, red-eyed vireos and ovenbirds (Frazer 1992) are less abundant near some forest edges, while hooded warblers, black-and-white warblers, (Frazer 1992) and cowbirds (O'Conner and Faaborg 1992) are more abundant near some forest edges. Reproductive success of some species is lower near even-aged regeneration cuts in some landscapes, but results are highly variable. Research in southern Indiana compared reproductive success of Acadian flycatchers, wood thrushes, red-eyed vireos, and worm-eating warblers between interior sites (heavily forested sites >5 km from agricultural habitats) and forest sites adjacent to clearcuts (D. Whitehead, unpubl. data). Across all species, parasitism was significantly greater near 4 of 5 clearcuts than in the interior site. Acadian flycatcher and red-eyed vireos were parasitized significantly more frequently near 4 of 5 clearcuts; wood thrushes near 1 of 5 clearcuts ($p<0.10$). Across all species, there were significantly higher proportions of depredated nests at 2 of the 5 clearcut sites; no significant differences were detectable at 3 sites. Species-specific comparisons indicate that the proportion of depredated Acadian flycatcher nests was significantly higher at 1 clearcut site (fig. 5).

Uneven-aged Methods

Uneven-aged management creates a wide range of tree sizes or ages within a stand; by definition stands should include a minimum of 3 age classes (Smith 1962). Under the single-tree selection method, individual trees are selected for harvest and under the group-selection method, small groups of trees from 0.04-0.2 ha are selected. The objective of selection cutting is to maintain a defined distribution of tree diameters. The diameter distribution is defined by a "q-value" that represents the ratio of the number of trees in consecutive diameter classes and essentially determines the steepness of the curve. Stands usually are harvested whenever stocking exceeds 80 percent, which on good sites should be every 15 to 20 years (Roach 1963, Law and Lorimer 1989, Marquis 1989). Group selection can successfully regenerate oaks. Single-tree selection tends to favor shade-tolerant species, although it has been used with

some success in maintaining oak forests in xeric ecosystems in the Missouri Ozarks (Johnson 1993).

Unlike even-aged methods, selection cutting does not produce large changes in habitat structure or turnover in bird communities at a stand level, but neither does it provide for the range of species at the landscape scale. Selectively cut stands retain most mature forest species. Selection cutting creates gaps and high vegetation diversity, which benefits many species. However, the selective removal of trees results in fewer large canopy trees and denser understories, which results in lower numbers of some species than in mature even-aged stands (fig. 4).

Canopy gaps resulting from the harvest of single trees or groups of trees provide habitat for a variety of migrant birds. Uneven-aged stands have a well-developed understory and subcanopy because of frequent canopy gaps. The presence of several well-developed vegetation levels and more complex habitat structure than even-aged stands could result in higher within-stand bird species diversity. Species such as the hooded warbler and indigo bunting appear to be able to make use of small gaps typical of single-tree selection, while other species such as yellow breasted chats, blue-winged warblers, and prairie warblers require large openings more typical of clearcuts (Annand and Thompson, in press) (fig. 4).

Landscape level impacts

Unlike even-aged management, selection cutting maintains a mature tree component at all times and does not create a mosaic of different-aged stands. This may benefit forest-interior warblers because large tracts of forest with mature trees can be maintained. However, if edge effects occur around group selection openings, they could cause local populations to become sinks, because while small, these openings can be numerous and widely dispersed throughout the forest (Thompson 1993). Selection cutting will not provide habitat for species that require larger openings or stands of early successional forest, or a diversity of even-aged stands.

Edge effects

Research in southern Indiana compared reproductive success of Acadian flycatchers, wood thrushes, red-eyed vireos, and worm-eating warblers between the forest interior and 2 sites that contained several group selection openings (D. Whitehead, unpubl. data) (fig. 5). Across all species, there were no significant differences in parasitism levels. Parasitism levels were significantly higher for the worm-eating warbler at one of the managed sites. Across all species, the proportion of depredated nests was significantly higher at one of the managed sites, but there were no significant differences for individual species. Ongoing research in southern Illinois suggests that species reproductive success is not strongly affected by selective cutting, although parasitism levels were higher for some species in recent cuts. Potentially higher parasitism levels could be the result of changes in forest structure as opposed to edge effects or fragmentation resulting from group openings. Reproductive success of some gap-dependent species (e.g., hooded warbler and white-eyed vireo, but not indigo bunting) was very low (S. Robinson, unpubl. data).

Comparison of Landscape-Level Impacts of EAM and UEAM

The effects of even-aged and uneven-aged regeneration methods must be considered at a landscape level to assess their impact on bird populations. Thompson (1993) used simulation modeling to compare the effects of group selection and clearcutting on a forest-interior bird population. Simulated population size declined up to 40 percent in a forest managed by clearcutting on a 100-year rotation, but projected populations differed only slightly in models with and without edge effects (fig. 6). This suggests that most of the decline was due to the conversion of older stands to younger stands (a reduction in habitat suitability or carrying capacity) because even-aged management did not create enough edge to greatly affect production, and that forest interior bird populations may persist at reduced levels in managed forests even with deleterious edge effects (Thompson 1993). In simulations without edge effects, group selection cutting had little impact on the modeled population because the total area in group openings was small and it was assumed that carrying capacity of the surrounding forest was unaffected. However, if edges around group selection openings function as ecological traps, they could result in local extirpations or population sinks, because while small, these openings could be much more numerous and widely

dispersed than those created by clearcutting. If openings created by selection cutting do not function as ecological traps, then selectively cut forests could maintain higher populations of a mature-forest species than forests managed by clearcutting (fig. 6) (Thompson 1993).

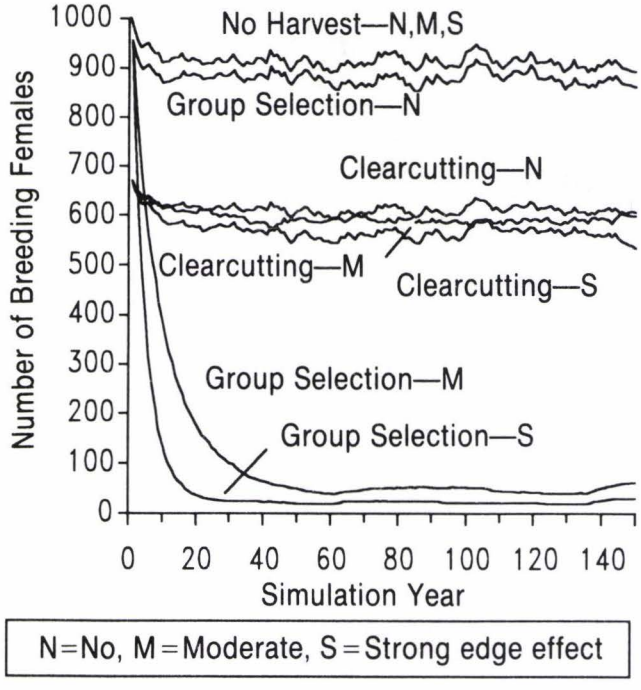


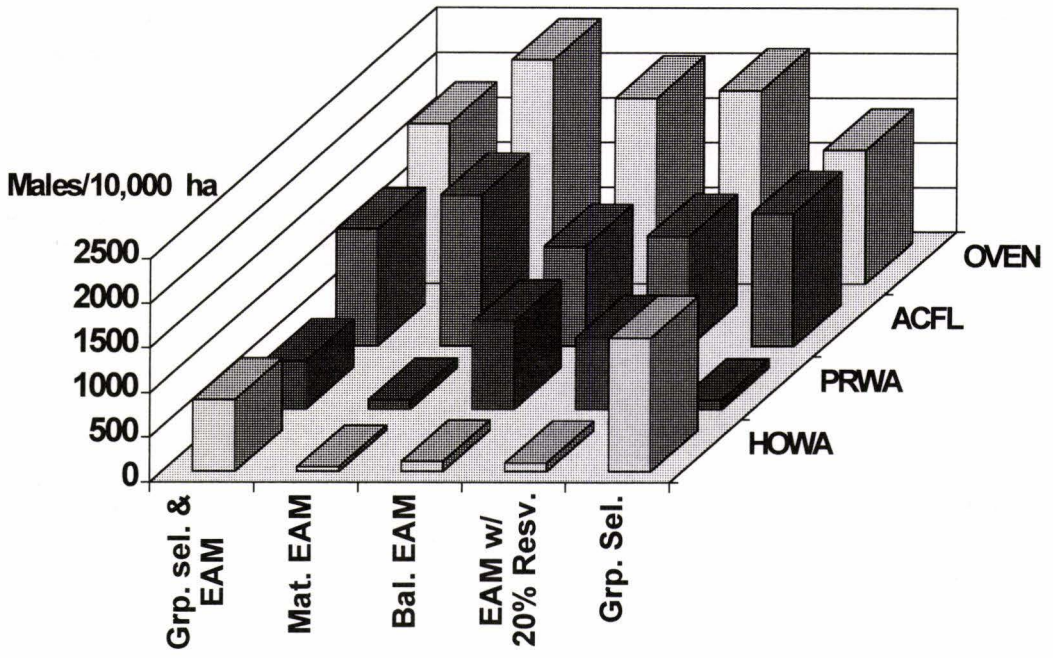
Figure 6.—*Simulated effects of clearcutting and selection cutting on a forest interior bird in a Central Hardwood forest (adapted from Thompson 1993).*

We used a simple model to compare the abundance of 4 species in Central Hardwood landscapes under 4 different management scenarios (fig. 7). We modeled abundance of a species that prefers small gaps (hooded warbler), a species that prefers large stands of young forest (prairie warbler), and two characteristic forest interior species that breed in mature forest (Acadian flycatcher and ovenbird). We used previously reported absolute and relative density estimates for these species in stands managed by silvicultural practices in the Missouri Ozarks (Thompson *et al.* 1992, Annand and Thompson 1997) to assign density values to different aged even-aged forest and to forest managed by a combination of group and single tree selection (fig. 7). We estimated the amount of each of these habitats in a 10,000-ha landscape to determine total population size. We modeled 5 management scenarios that represent realistic alternatives for public and private lands in the Central Hardwood Region. The mature even-aged management scenario was a landscape composed entirely of mature (>60-year-old) forest. The balanced even-aged management scenario was a landscape managed by the clearcut method with a 100-year rotation and balanced age class distribution (i.e., 10 percent 0- to 10-year-old forest, 10 percent 11- to 20-year-old forest, etc.). Even-aged management with 20 percent reserved was managed as the even-aged landscape except 20 percent of the area was reserved from harvest and was always mature forest. The group selection method was based on recommendations for Central Hardwood forests by Law and Lorimer (1989). The group selection and EAM option consisted of equal areas managed by the clearcut method and the selection method. The model was based on density data from managed forests and the area of each forest habitat in the modeled landscape; it accounts for edge effects on abundance and reproductive success only to the extent these effects were reflected in the original density estimates by Thompson *et al.* (1992) and Annand and Thompson (1997).

Species abundances under different management scenarios varied as a function of their habitat preferences (fig. 7). For instance, ovenbirds were most abundant in the Mature EAM landscape because they are most abundant in mature even-aged forest. Prairie warblers were most abundant in the Balanced EAM landscape because they prefer young forest stands. More importantly the model allows comparison of densities among species and management options. From this perspective, the only option that maintains moderate to high densities of all 4 species is a mix of EAM and group selection. While this model represented a single landscape, it is not important (or necessarily always desirable) that this mix be provided within one landscape; it could also occur among a group of landscapes.

Forest-Wildlife Openings

Forest wildlife openings are small openings created in the forest for a variety of objectives, including habitat for some wildlife and plants as well as recreational opportunities including hunting and wildlife observation. These openings are managed in various ways, including planting them to cereal grains, grass and



Bird abundance in habitats created by forest management

Species	Territories / 10 ha				
	Forest age (even-aged)				uneven-aged
	0-10	10-20	20-40	>40	(group selection)
HOWA	0.3	0.3	0.1	0.1	1.5
PRWA	5.5	3.5	0.1	0.1	0.1
ACFL	0.0	0.1	0.5	1.7	1.5
OVEN	0.3	1.5	2.0	2.5	1.5

Figure 7.—*Simulated bird abundance in central hardwood landscapes under different management scenarios.*

forbs, or maintaining them in a grass or shrub stage of succession by disking, mowing, or fire. They provide habitat for some resident songbirds and NTMB's such as indigo buntings but often are too small for early successional NTMB's such as yellow-breasted chats, blue-winged warblers, and prairie warblers.

Because of their small size, these openings do not result in the loss of significant amounts of mature forest. Of greater potential concern are possible impacts of edge effects on mature-forest birds in adjacent forest stands. In southern Indiana, Whitehead *et al.* (unpubl. data) found higher parasitism levels across all species, and individually for the wood thrush, red-eyed vireo, and indigo bunting, in sites with forest openings compared to forest interior sites. Similarly the proportion of depredated nests was greater for all species combined, and individually for the wood thrush, at sites with openings (fig. 5).

Fire and Neotropical Migratory Birds

Recognition of the need for periodic disturbance has led to increased use of prescribed fire as a management and restoration practice in oak-dominated ecosystems, but its effect of this trend in restoration on forest birds is not fully understood. Clearly, an effort as potentially influential as savanna restoration will strongly effect certain species. As with other forest management practices, certain species of neotropical migrants likely will benefit from fire and savanna restoration and others will be adversely affected (fig. 4).

In addition to the forest birds associated with savannas that are identified in figure 4, the following species also have significant habitat associations with savannas: black-billed cuckoo, whip-poor-will, ruby-throated hummingbird, least flycatcher, purple martin, barn swallow, northern rough-winged swallow, cliff swallow, bank swallow, warbling vireo, dickcissel, and the orchard and northern oriole (J. Brawn, unpubl. data). Most of these species are declining in the region (Appendix 2). This list reflects a broad definition of a savanna as an area with 20- to 70-percent-canopy closure and a permanent herbaceous groundlayer (note that definitions differ widely among states and various agencies). Our criteria on inclusion on the list was that a species have an important habitat association with savannas and open woodlands during the breeding season. Certain species also have important associations with other habitats and ecosystems such as grasslands, shrublands, or forest (fig. 4). Importantly, many other species of permanent residents and short-distance migrants also are associated with midwestern savannas (Brawn 1994). The list of long-distance migrants is diverse in taxonomy and ecology and includes species as different as whip-poor-wills and summer tanagers. One life history trait common to many of these species is that they forage on the wing and catch insects in the air. Obviously, the ability of birds to forage this way is favored by an open canopy.

The majority of species associated with savannas are significantly declining within the region (Appendix 2). Very few have significantly increased. Therefore, the conservation status of these birds as a group is worrisome. Fire and restoration will provide more habitat for the subset (a majority) of these species that are not equally abundant in non-disturbed areas. We don't know what effects fire will have on ecological factors that affect reproductive success; specifically, rates of cowbird parasitism and nest predation. Preliminary evidence from the fragmented landscapes of Illinois suggests that fire does not significantly affect productivity, but reproductive success is so low in these landscapes that effects would be difficult to demonstrate (J. Brawn, unpubl. data). Savanna restoration in heavily forested landscapes may yield more positive results because of the effects of forest fragmentation on reproductive success (Donovan *et al.* 1995a, Robinson *et al.* 1995). The influence of changes in habitat due to fire will depend on the landscape context of restoration efforts. A large burn unit surrounded by large holdings of prairie or closed-canopy forests likely will be more beneficial to most birds than a similar burn in a landscape dominated by agriculture.

Other considerations for prescribed fire pertain to factors under the control of local managers. When and how often to burn are important questions and no single prescription will be valid for all areas. While restoration efforts are underway, burning will be relatively frequent. Once savanna-like conditions are established with respect to canopy closure and a herbaceous groundlayer, burning may become less frequent. Importantly, burning is only one element of savanna or oak woodland restoration. Other techniques such as planting and mechanical manipulation of vegetation may also be required (see Botts *et al.* 1994 for a full discussion of restoration techniques and considerations).

Finally, although we view prescribed fire as an important management option for the Central Hardwoods, we do not believe that sound conservation will be served by burning "everywhere." Species that nest in dense understory or shrubs, such as the wood thrush and the hooded warbler, will not benefit from fire (Brawn, unpubl. data). Close attention should be given to the types of areas that burned naturally and those that did not burn. An effective approach to this problem is the concept of "landscape burns," whereby burns are allowed to proceed where they may, but are not forced into areas (e.g., ravines) that are too mesic to have burned naturally.

Another use of prescribed fire is the management of edges. Historically, savannas often were juxtaposed between prairies and wooded habitats. Thus, there may have been a gradual transition between non-wooded and wooded habitat. In modern midwestern landscapes, sharp transitions between agriculture and forest may create edges that are highly unfavorable for neotropical migrants (Suarez *et al.*, in press). Fire may soften the transition and create edges that are more hospitable as breeding habitat.

CONSERVATION STRATEGIES

Management Issues for the Conservation of Forest Birds In the Central Hardwood Region

We believe two dominant issues concern conservation of migrant birds in the Central Hardwood Region:

1. Land use patterns have resulted in great variation in the amount of forest cover. We have strong evidence that NTMB populations are structured as sources and sinks, but we do not know how populations are linked by dispersal. This question is important because land management actions in one population can affect other populations. For example, fragmentation of source habitat could lead to extinction of sink populations, or abundant sink habitat could drain source populations (Donovan *et al.* 1995b). Conservation planning must recognize the spatial structure of populations in the Midwest and emphasize enhancement and preservation of regional source populations.

2. Conservation of NTMB's is competing with other land uses for a limited land base. There may be conflicts with commodity production, recreation, wilderness management, and wildlife management; among NTMB species; or between species and ecosystem management. Nevertheless, we have assessed costs and benefits of these practices for NTMBs but acknowledge compromises will be required for multi-resource planning.

Address Landscape-Level Fragmentation at a Regional Scale

The Central Hardwood Region consists of a mix of source and sink populations whose status is linked to landscape-scale fragmentation of forest. The population status of forest-dwelling migratory birds is related to the amount of source and sink habitat in the region. Given the current concern for migratory birds and observed declines in some species, the first step should be to ensure that sufficient source populations exist in the region. Only a few large, heavily forested landscapes remain in the Central Hardwood Region (fig. 2). We believe a prudent conservation strategy for the region is to maintain a high percentage of forest cover in these landscapes, to maintain the largest forest tracts in the region, and to take advantage of any opportunities to restore and recreate larger forest tracts in more fragmented landscapes (the "macrosite" approach mentioned in Robinson *et al.* 1995). Most land in the Central Hardwood Region is privately owned. Extensively forested land, however, is disproportionately owned by Federal and State agencies (fig. 2). The relationship of public lands to the distribution of forest cover greatly enhances the value of public land to forest bird conservation. It is not coincidental that the core of nearly all the large tracts of forest in the Central Hardwood Region are National and State Forests. Federal and State agencies have long protected, conserved, and in some cases reforested these lands; continued forest conservation by the agencies is critical to forest breeding migratory birds. Maintaining a balance of birds in source and sink habitat may be crucial to the long-term survival of regional populations (Urban and Shugart 1986, Temple and Cary 1988, Donovan *et al.* 1995b).

Provide Appropriate Habitats and Landscape Pattern

In addition to protecting extensively forested landscapes, appropriate forest habitats must be provided within these landscapes. Within Central Hardwood landscapes migratory birds breed within a wide range of forested and semi-forested habitats. Even when we reviewed habitat use by a reduced set of priority species, those species still spanned a wide range of habitats. Conservation planning is further complicated when we acknowledge that many of these lands are managed for multiple use, including recreation and forest products.

Plan forest management activities to meet habitat needs

Priority species, and forest birds in general, require a variety of forest habitats. Land

management planning should provide these habitats across the region. Land managers and the public must recognize that this means that the management of any particular landscape will benefit some species but will have negative effects on others, but that across landscapes the needs of all species will be met. We believe a mix of even- and uneven-aged silvicultural practices, designated reserve areas, and use of prescribed fire will be required within the Central Hardwood Region to meet bird conservation objectives and other objectives for forest lands. We used a simple model based on landscape composition to demonstrate that almost any mix of regulated, sustainable forest management practices likely will sustain most species across a landscape; densities, however, differed greatly among alternatives. Land management objectives for other resources also will influence the balance of managed forest lands and reserved areas. Ultimately, conservation planning for migratory birds should develop minimum habitat requirements and plan the distribution of these within a region. This approach is currently being pursued by the Partners in Flight program.

Reduce internal edge and disturbance where appropriate

Reducing the amount of non-forest habitat in a landscape, and the edges it creates, should benefit forest breeding birds by creating more breeding habitat and reducing nest depredation rates and brood parasitism. These habitats include cropland, grassland, pasture, mowed rights-of-way, and suburban, urban, or commercial developments. These habitats likely increase the number of predators and cowbirds in the landscape. Reforestation, or even less drastic modifications of land-use, on small non-forested inholdings in large forest tracts could have disproportionate benefits to forest breeding birds.

We had difficulty developing recommendations concerning disturbances and edges resulting from forest management. Some priority species within the Central Hardwood Region require disturbance-dependent communities, such as young forest, glades, barrens, or savannas. The significance of deleterious edge effects associated with forest management differs among species and sites. The only evidence of population impacts is from simulation modeling, which suggests edge effects can result in population declines and even extirpations (Temple and Cary 1988, Thompson 1993), but that potential edge effects from regulated forest management are tolerable (Thompson 1993). We recommend reducing disturbance and edges from forest management practices where late-successional, forest-interior species are a priority, while acknowledging that edge is a necessary byproduct of desirable management practices where early successional species or other forest management objectives require disturbance. Where possible, promote gradual, shrubby edges rather than abrupt edges, and keep intensive recreation development out of the core of some large forest tracts. Depending on forest ownership, legal mandates, historical landscape composition, and management objectives, practices that create early successional habitats or produce forest products will be appropriate. The challenge in these circumstances is to meet habitat or commodity objectives while not creating unnecessary edge effects. Again, as mentioned in the previous section, a mix of managed forest lands and reserved areas within an adequate forest landbase will accommodate the needs of Central Hardwood NTMBs.

Plan comprehensively across ownerships

Land management planning for the conservation of forest-dwelling NTMB's must occur across public and private lands. An example of comprehensive planning across ownerships with voluntary participation by private landowners is the Coordinated Resource Management Program in Missouri (Missouri Department of Conservation). Because conservation agencies have more direct control over public lands, and within the Central Hardwood Region they form the core of most heavily forested landscapes, they have unique value. NTMB populations should benefit if public lands are managed to complement private lands and to provide disproportionate amounts of high-quality, rare ecosystems such as savannas and glades, old-growth forest, and large heavily forested landscapes. In landscapes in which private lands provide abundant early-successional habitat, manage for disproportionate amounts of mature forest on public lands or the use of uneven-aged management instead of even-aged management. Alternatively, where private land provides too little early-successional habitat, or unsuitable early-successional habitat (such as from partial-cut methods as high-grading), public lands may need to provide shrublands and clearcuts in heavily forested

landscapes where reproductive success is likely to be high. Opportunities for restoration of natural communities, such as savannas, will most likely be greatest on public lands.

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Appendix 1.—*Migrant landbirds that breed in Illinois, Indiana, Missouri, and Ohio. Species followed by 'A' are long-distance migrants and winter south of the United States. Species followed by 'B' winter primarily in the Unites States, but part of the population winters south of the United States. List developed by the Partners in Flight program in cooperation with Colorado Bird Observatory and State conservation agencies.*

Turkey vulture	B	Purple martin	A	Prothonotary warbler	A
Osprey (supp.)	B	Tree swallow	B	Worm-eating warbler	A
American swallow-tailed kite	A	Northern rough-winged swallow	A	Swainson's warbler	A
Mississippi kite	A	Bank swallow	A	Ovenbird	A
Northern harrier	B	Cliff swallow	A	Northern waterthrush	A
Sharp-shinned hawk	B	Barn swallow	A	Louisiana waterthrush	A
Cooper's hawk	B	Brown creeper	B	Kentucky warbler	A
Red-shouldered hawk	B	House wren	A	Mourning warbler	A
Broad-winged hawk	A	Sedge wren	B	Common yellowthroat	A
Swainson's hawk	A	Marsh wren	B	Hooded warbler	A
Red-tailed hawk	B	Golden-crowned kinglet (supp.)	B	Canada warbler	A
American kestrel	B	Blue-gray gnatcatcher	A	Yellow-breasted chat	A
Peregrine falcon	A	Eastern bluebird	B	Summer tanager	A
Killdeer	B	Veery	A	Scarlet tanager	A
Upland sandpiper	A	Hermit thrush	B	Rose-breasted grosbeak	A
Mourning dove	B	Wood thrush	A	Blue grosbeak	A
Black-billed cuckoo	A	American robin	B	Indigo bunting	A
Yellow-billed cuckoo	A	Gray catbird	A	Painted bunting	A
Burrowing owl	A	Northern mockingbird	B	Dickcissel	A
Long-eared owl	B	Cedar waxwing	B	Rufous-sided towhee	B
Short-eared owl	B	Loggerhead shrike	B	Chipping sparrow	A
Common nighthawk	A	White-eyed vireo	A	Clay-colored sparrow	A
Chuck-will's-widow	A	Bell's vireo	A	Vesper sparrow	B
Whip-poor-will	A	Solitary vireo	A	Lark sparrow	A
Chimney swift	A	Yellow-throated vireo	A	Lark bunting	A
Ruby-throated hummingbird	A	Warbling vireo	A	Savannah sparrow	B
Belted kingfisher	B	Red-eyed vireo	A	Grasshopper sparrow	A
Yellow-bellied sapsucker	B	Bachman's warbler	A	Song sparrow	B
Red-naped sapsucker	B	Blue-winged warbler	A	Swamp sparrow	B
Northern flicker	B	Golden-winged warbler	A	Bobolink	A
Eastern wood-pewee	A	Northern parula	A	Red-winged blackbird	B
Acadian flycatcher	A	Yellow warbler	A	Eastern meadowlark	B
Alder flycatcher	A	Chestnut-sided warbler	A	Western meadowlark	B
Willow flycatcher	A	Magnolia warbler	A	Yellow-headed blackbird	A
Least flycatcher	A	Black-throated blue warbler	A	Brewer's blackbird	B
Eastern phoebe	B	Black-throated green warbler	A	Brown-headed cowbird	B
Great crested flycatcher	A	Blackburnian warbler	A	Orchard oriole	A
Western kingbird	A	Yellow-throated warbler	A	Northern oriole	A
Eastern kingbird	A	Prairie warbler	A	Purple finch	B
Gray kingbird	D	Cerulean warbler	A	Pine siskin	B
Scissor-tailed flycatcher	A	Black-and-white warbler	A	American goldfinch	B
Horned lark	B	American redstart	A		

Appendix 2.—*Trends in abundance of forest birds, based on the Breeding Bird Survey for years 1966-1994[1], for four states in the Central Hardwood Region.*

Species	ILLINOIS			INDIANA			MISSOURI			OHIO		
	Trend[2]	# Rts[3]	No/Rt[4]	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt
Turkey vulture	1.5	22	0.37	2.2	28	1.63	4.8***	40	2.18	4.70**	56	2.41
Wild turkey							11.5	24	0.49			
Cooper's hawk										3.2	21	0.06
Red-tailed hawk	7.7**	65	0.70	8.5***	39	0.76	-1.0	45	1.08	3.1	57	0.71
Mourning dove	-0.2	80	33.42	-0.1	44	46.82	-2.7***	45	38.91	0.6	69	36.63
Black-billed cuckoo	-2.1	33	0.24	-2.6	23	0.22				-6.7*	32	0.26
Yellow-billed cuckoo	-3.4***	73	3.74	-2.8**	41	2.92	-0.4	45	10.15	-3.0***	53	1.09
Great horned owl	7.7	33	0.15	9.0*	18	0.15	-1.4	25	0.45			
Barred owl	-5.0	14	0.08				4.7	18	0.12			
Chuck-will's-widow							0.7	25	1.23			
Whip-poor-will	-16.1**	16	0.25				-0.8	32	1.40			
Chimney swift	-1.7***	77	14.52	-1.6*	43	12.72	-0.9	45	9.30	-0.4	68	15.24
Ruby-throated hummingbird	-2.0	28	0.25	0.5	26	0.50	-5.7	29	0.51	2.5	42	0.38
Red-headed woodpecker	-1.4*	75	5.25	-2.8***	42	3.24	-0.3	44	6.04	-5.3***	49	1.10
Red-bellied woodpecker	1.4	69	3.88	-0.2	43	5.00	-0.7	44	7.14	0.8	61	2.44
Downy woodpecker	1.0	64	1.47	-0.8	43	2.34	0.6	45	2.54	0.4	64	1.79
Hairy woodpecker	2.6	32	0.17	-0.3	25	0.25	-2.8	35	0.41	-6.5*	33	0.15
Northern flicker	-2.8***	77	3.26	-2.2***	44	4.10	-1.3*	45	3.33	-3.2***	66	3.17
Pileated woodpecker	7.2*	15	0.22	1.8	16	0.65	-1.0	27	0.91	2.3*	30	0.61
Eastern wood-pewee	-0.7	75	2.46	-0.6	40	5.17	-1.8	44	6.13	-1.8***	66	3.76
Acadian flycatcher	-2.9	24	0.52	-1.5	26	1.83	2.4	21	0.37	-1.4	42	2.11
Eastern phoebe	4.8***	59	0.64	2.7**	30	0.88	3.2**	44	3.45	-0.3	53	1.77
Great crested flycatcher	-1.4**	70	1.95	-0.3	41	2.33	-1.0	45	4.94	-2.8***	64	1.65
Blue jay	-0.9	80	9.75	-0.2	44	9.44	-0.5	45	17.02	0.8	69	8.92
Black-capped chickadee	3.1***	39	1.20				0.0	26	2.96	3.8***	14	1.37
Carolina chickadee	-0.5	22	0.79	0.0	33	2.15	-0.7	22	3.91	1.8***	45	2.13
Tufted titmouse	1.6*	68	4.12	1.7**	44	8.60	-0.8	45	11.30	-1.2*	69	8.71
White-breasted nuthatch	5.0***	51	0.89	0.8	42	1.84	3.1***	39	2.87	2.8	63	1.21

Appendix 2. Continued.

Species	ILLINOIS			INDIANA			MISSOURI			OHIO		
	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt
Carolina wren	5.1*	42	1.77	6.7***	34	2.78	0.6	36	2.15	-1.0	56	2.51
Bewick's wren							-2.5	23	1.04			
House wren	1.6**	72	6.24	2.8***	42	7.08	-2.0***	42	4.91	2.9***	67	9.62
Blue-gray gnatcatcher	5.9	34	0.80	0.7	26	1.24	-0.2	36	3.85	-0.5	44	1.79
Wood thrush	-1.5	50	0.93	-1.5**	42	3.37	-1.5	29	1.38	1.4*	63	5.96
American robin	3.2***	80	51.61	3.0***	44	59.85	2.3***	45	32.58	3.6***	69	60.51
Gray catbird	1.5**	74	3.53	1.2**	44	6.26	-2.2**	45	2.94	1.4***	69	7.11
Northern mockingbird	-3.4***	61	4.48	1.8	37	4.44	-3.0***	45	10.39	0.9	47	3.17
Brown thrasher	-0.7*	80	6.16	-0.7	44	4.51	-1.6***	45	8.94	-1.4*	64	2.64
Cedar waxwing	14.3***	55	1.46	7.8***	40	3.00				2.1	58	3.99
White-eyed vireo	-0.9	24	0.96	0.6	25	1.29	0.5	26	1.68	2.8**	40	1.71
Bell's vireo	-3.7	22	0.16				-3.2	22	0.58			
Yellow-throated vireo	4.2	27	0.27	0.7	27	0.99	-2.1	27	0.37	0.00	49	1.03
Warbling vireo	1.2	68	2.01	0.6	43	3.95	1.0	38	1.41	-3.00**	58	1.81
Red-eyed vireo	-2.7**	53	0.61	0.2	39	3.81	2.4	34	3.20	1.2**	66	6.43
Blue-winged warbler										-0.8	32	0.94
Northern parula	6.7**	16	0.19				0.3	33	1.24			
Yellow warbler	6.1***	49	0.54	2.4**	39	2.09				0.2	64	5.16
Yellow-throated warbler				4.8***	14	0.48				-1.5	17	0.30
Prairie warbler							-6.3	14	0.97	-1.2	24	3.15
Cerulean warbler										-4.8***	30	0.47
Black-&-white warbler							2.6	18	0.15			
American redstart										-3.6*	23	0.73
Ovenbird										0.2	25	1.84
Louisiana waterthrush							3.8	14	0.12			
Prothonotary warbler	1.2	14	0.09									
Kentucky warbler	-2.8	14	0.29	1.1	14	1.56	3.6**	29	0.95	0.00	23	0.97
Common yellowthroat	-0.4	77	9.75	-0.6	43	14.72	-0.7	45	9.89	1.4**	69	10.94

Appendix 2. Continued.

Species	ILLINOIS			INDIANA			MISSOURI			OHIO		
	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt	Trend	# Rts	No/Rt
Hooded warbler										3.7	20	0.74
Yellow-breasted chat	-4.1***	44	1.65	-1.9**	33	3.54	-0.8	39	4.69	-2.8***	45	3.68
Summer tanager	-3.5	15	0.34	-2.1*	15	0.91	0.2	37	3.02	0.4	20	0.34
Scarlet tanager	-1.5	30	0.22	2.7**	33	1.31	8.5	20	0.33	1.2	50	1.87
Northern cardinal	1.0	80	18.83	0.1	44	24.43	-1.1**	45	29.05	1.2**	69	22.53
Rose-breasted grosbeak	3.3***	53	1.53	1.9	21	0.44	0.4	24	1.63	-1.6	26	0.73
Blue grosbeak	1.6	16	0.33				1.0	40	3.59			
Indigo bunting	-0.8*	80	21.96	-1.6***	44	34.20	-0.1	45	25.24	1.1***	69	19.06
Rufous-sided towhee	-0.8	60	1.53	0.4	38	3.24	-2.0*	42	4.84	-1.4**	62	3.97
Chipping sparrow	0.4***	75	5.04	2.8**	44	10.45	-0.5	41	3.66	1.8**	69	11.95
Brown-headed cowbird	2.1**	80	13.27	0.3	44	12.49	0.2	45	24.92	-2.1***	69	9.62
Orchard oriole	-8.9	48	0.97	-0.1	36	1.69	-0.6	44	2.92	0.8	41	0.45
Northern oriole	1.1	74	2.97	1.8*	43	2.96	0.1	41	3.57	1.2	64	2.61

[1]Provided by Bruce G. Peterjohn, National Biological Service, Patuxent Environmental Science Center, Laurel, Maryland

[2]Percent annual change. Asterisk indicate statistical significance; ***,0.01; **, 0.05; *, 0.1.

[3]Sample size for trend analysis.

[4]Mean number of birds per BBS route

MANAGEMENT OF WESTERN GREAT LAKES FORESTS FOR THE CONSERVATION OF NEOTROPICAL MIGRATORY BIRDS

R. W. Howe[1], G. Niemi[2], and J. R. Probst[3]

ABSTRACT.—Forests and associated wetlands in the Great Lakes region support some of North America's most diverse assemblages of breeding birds. The majority of species in the western Lake States (as in most ecological communities) are uncommon and localized in occurrence. Many of the region's bird species show clear associations with habitat, but landscape characteristics and geographic factors also affect species' distributions and abundances. Local field surveys provide a mechanism for establishing species priorities for individual management units. Effective conservation plans for most species must be empirically based; in other words, managers need to know where priority species occur and where they are most productive. Management of habitat is a first step for bird conservation, but species interactions and biogeographic factors inevitably complicate simple management prescriptions. We recommend a management scheme that integrates traditional habitat management prescriptions, landscape-level planning, and continuous monitoring of local bird populations.

INTRODUCTION

Northern parts of the western Lake States (Minnesota, Michigan, and Wisconsin) encompass some of the most extensively forested landscape in the Midwest. The continuity of forest and the diversity of wetlands, lakes, and forest types makes this region particularly significant for neotropical migratory birds (Temple *et al.* 1979). We reviewed the ecological characteristics of forest landscapes in the western Lake States and their relevance to forest management and neotropical migratory birds. Although much remains to be learned, much is known about the bird populations in this region. Empirical findings provide opportunities for enlightened forest management that are unavailable in many other regions. Field evidence reveals that bird distributions and habitat interactions in northern forest landscapes are complex. We argue that simple prescriptions for bird conservation in these forests do not exist. Reliable field information, however, helps managers set species priorities and formulate specific conservation plans. The key to successful efforts, we maintain, is an ongoing, stepwise assessment of bird species and other resources at several levels of resolution (e.g., Probst and Thompson 1996), culminating in carefully targeted surveys to achieve a reliable information base that is actively incorporated into forest management activities.

Western Great Lakes Forests

The Upper Midwest is characterized by a continental ecotone between grasslands and mostly forested landscapes which extend into Canada. The region described here (fig. 1) corresponds with the western portion of the Laurentian Mixed Forest Province of Bailey (McNab and Avers 1994, see fig. 2 in Probst and Thompson 1996). Climatic and ecological gradients extending from the southwest to the northeast (Curtis 1959), together with complex glacial topography (Paul and Paul 1990), produce a forest landscape that is transitional in character over a broad geographic area. Extensive areas of spruce-fir, upland conifers,

[1] *Department of Natural and Applied Sciences, University of Wisconsin-Green Bay, Green Bay, WI 54311-7001 (414) 465-2272.*

[2] *Natural Resources Research Institute, University of Minnesota, Duluth, MN 55811 (212) 720-4270.*

[3] *Forestry Sciences Laboratory, 5985 Highway K, Rhinelander, WI 54501 (715) 362-1156.*

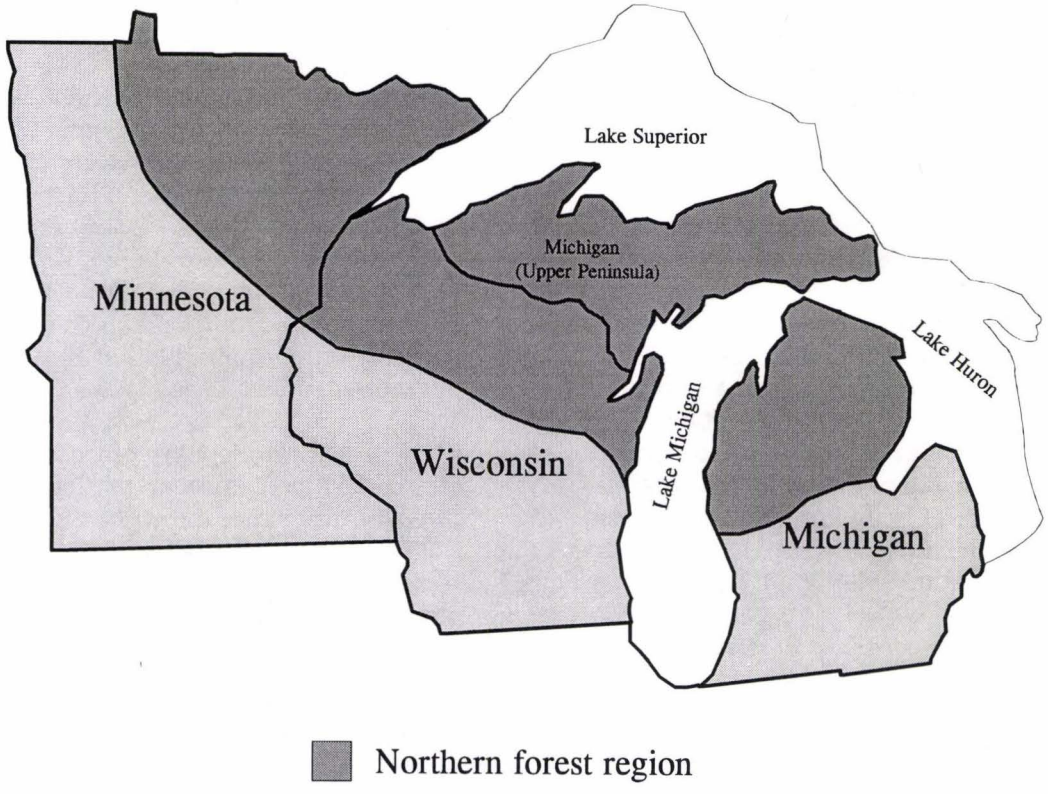

Figure 1.—*Distribution of western Great Lakes forests, corresponding to the western section of Bailey's (1994) Laurentian Mixed Forest Province.*

and northern hardwoods are associated with finer ecological divisions of the region (Albert, in press). Minnesota has the highest proportion of spruce-fir forests, but the lowest proportion of northern hardwoods. Michigan has the largest area of northern hardwoods and pine forest.

Mesic northern hardwood forests dominated by sugar maple (*Acer saccharum*), with varying amounts of beech, hemlock (*Tsuga canadensis*), yellow birch (*Betula lutea*), green ash (*Fraxinus pennsylvanica*), and basswood (*Tilia americana*) occupied approximately 47 percent of the presettlement forest landscape in the western Lake States (Frelich 1995). Other major forest categories included swamp conifers (13 percent), red/white pine (12 percent), spruce-fir-birch (10 percent), oak-hickory (9 percent), riverbottom forest (6 percent), and jack pine (4 percent). Frelich estimated that 89 percent of the original northern hardwood forests and 68 percent of all forests in this region existed as old growth.

Studies of remnant old-growth forests by Mladenoff and Pastor (1993) demonstrate that northern forest landscapes are complex and highly heterogeneous, rendering current forest classification and forest type delineation somewhat limited and possibly misleading for broader applications. Davis (1983) has shown that the geographic ranges of hemlock, beech, and other species are still expanding in response to post-glacial environmental changes. Hardwood forests of northeastern Wisconsin, for example, are richer in species than those of northwestern Wisconsin because beech, yellow birch and hemlock decrease in abundance along a moisture gradient from east to west (Curtis 1959). For reasons such as this, ecologists are devising multi-factional Ecological Classification Systems (ECS) that nest and network biotic distributions in a biophysical hierarchy (McNab and Avers 1994).

Pre-settlement Ecology / Disturbance Regimes

Pre-settlement landscapes of the western Great Lakes region were dynamic as a result of natural disturbances and ecological succession (Stearns 1990, Frelich and Lorimer 1991, Mladenoff and Pastor 1993). Fire was most important on pine-dominated uplands and in boreal environments, where disturbance intervals were in the order of 100 years (Heinselman 1973). Large-scale disturbances in hardwood-hemlock forests were less frequent (> 1000 years), most often the result of windthrows (Canham and Loucks 1984). Canopy openings caused by small, local windthrows represented the most common source of disturbance (Frelich and Lorimer 1991). Outbreaks of insect herbivores, disease, and browsing by large mammals also played roles in these natural disturbance regimes (White and Mladenoff 1994).

Natural forest succession in northern forest landscapes involves a variety of tree species combinations, with aspen (*Populus tremuloides* and *P. grandidentata*), birch (*Betula papyrifera* and *B. populifolia*), balsam fir (*Abies balsamea*), and pin cherry (*Prunus pensylvanica*) prominent in mesic sites after disturbance. Alder, black spruce, and tamarack are typical of early-successional wetland sites, while jack pine, red pine, and oaks occur in open or frequently disturbed dry sites (Curtis 1959, Temple *et al.* 1979). Later-successional mesic forests include hardwoods (sugar maple, beech, basswood, yellow birch, white ash (*Fraxinus americana*) and ironwood (*Ostrya virginiana*)), white pine (*Pinus strobus*) and hemlock. Northern white-cedar (*Thuja occidentalis*), balsam fir, American elm (*Ulmus americana*), and black ash (*Fraxinus nigra*) dominate wet sites, while northern red oak (*Quercus borealis*) and red maple (*Acer rubrum*) are characteristic of drier sites. Just as local conditions affect the old growth forest composition in this transitional region, successional patterns are complicated by climate, soils, topography, and the nature of disturbance.

History of Management and Land Use

The naturally dynamic nature of this landscape has been further complicated by dramatic alteration of pre-settlement forests since the late 1800's (Stearns 1990). Frelich (1995) reports that only about 1.1 percent of the presettlement primary forest of the western Lake States remains in a condition that has not been significantly modified by humans. About 19.8 million ha of forest currently exists in the northern forest region, compared with 32.7 million ha during presettlement times. Using unlogged landscapes in the Sylvania Wilderness (6,070 ha) and the Porcupine Mountains (14,164 ha) of Upper Michigan as reference, Frelich estimates that 90 percent of the presettlement northern hardwoods forests consisted of old growth, dominated by trees 120 years old or older; today, only 9.7 percent of forests near the Sylvania Wilderness and Porcupine Mountains reach this age. Only 0.2 percent of the original forest remains unlogged or unburned as a result of human activity.

The species composition of this forest also has changed since European settlement. Notably, the extent of aspen-birch has increased from 1.4 percent to 22.6 percent; upland spruce-fir from nearly 0 (in this northern hardwoods region) to 10.5 percent; and pine-dominated forests from 1.4 percent to 5.0 percent. The species composition of northern hardwoods forests (which have been reduced in extent from 97.1 to 61.2 percent of the landscape) also has been altered by selective elimination of hemlock, white pine and other less prominent species. Temple *et al.* (1978) concluded that today northern forests of the Midwest are more homogeneous locally than the forests in which the resident birds evolved, a view that is supported by the more recent investigations of Frelich and Lorimer (1991), White and Mladenoff (1994) and others.

Conifer forests, particularly those dominated by swamp conifers, have experienced comparatively less modification in this landscape than have hardwood forests. Nevertheless, only 3 percent of the original area of jack pine forest, 0.6 percent of the red-white pine forest, and 4.4 percent of the swamp conifer forest remains unmodified by logging. Suppression of fire has led to further changes, even in the wilderness landscape of the Boundary Waters Canoe Area in northeastern Minnesota, the largest single block of primary forest in the Lake States (Frelich 1995). As a result of lower fire frequency, the area of aspen-spruce-fir-birch forest has increased at the expense of forests dominated by red and white pine. Mladenoff and Pastor (1993) argue that destructive logging, post-logging fires, and other human influences have "made recovery to any semblance of the presettlement state impossible."

Current Forest Conditions and Ownership Patterns

Total forest acres are similar among the three Lake States, but are highest in Michigan (table 1). More than half of Minnesota's forests are publicly owned, approximately twice the proportion in Wisconsin. Altogether more than 38 percent of the forest lands in the Lake States are publicly owned, and a significant fraction of the remainder is managed as private industrial forest. The government agencies responsible for forest management, however, vary significantly among the three states (table 1).

Most of the public forests in the Lake States are managed for multiple uses; in general, the more locally administered the forest (e.g., county vs. federal), the more intensive the management policies in terms of rotation age and harvest levels. Management of national forests, however, has probably been most controversial (Alverson *et al.* 1988, Kuhlmann 1990, Alverson *et al.* 1994). According to Frelich (1995), only 0.8 million ha (approximately 4 percent) of today's forest is reserved from logging, mainly in federally designated wilderness areas or their state equivalents, including the Boundary Waters Canoe Area Wilderness in northeastern Minnesota, Isle Royale National Park in Michigan, Porcupine Mountain State Park in Upper Michigan, and the Sylvania Wilderness of the Ottawa National Forest.

BIRDS OF WESTERN GREAT LAKES FORESTS

General Description of the Avifauna

Like the forest itself, the avifauna of the Upper Midwest is transitional and ecologically complex (Temple *et al.* 1979). Species assemblages in this region have varied in space and time since the retreat of the Wisconsin glacier 10,000 years ago (Davis 1983, Graham 1986). Nevertheless, a core assemblage of bird species occurs today throughout virtually all forested landscapes in the western Lake States. Systematic surveys using similar methods in the Nicolet National Forest (Howe *et al.* 1995), Chequamegon National Forest (Hawrot *et al.* 1994) and Superior and Chippewa National Forests (Hawrot *et al.* 1994) have identified the same common species (table 2). Forest composition and bird species differ among state, county, and privately owned forests, but ovenbird, red-eyed vireo, and several other familiar bird species (least flycatcher, black-throated green warbler, chestnut-sided warbler, Nashville warbler, hermit thrush) can be found commonly across most of the western Great Lakes forest region.

As in most species assemblages (Preston 1962, Brown *et al.* 1995), the majority of species inhabiting western Great Lakes forests are uncommon or rare (fig. 2). Howe *et al.* (1995) showed that only about 20 percent of the 150+ species in the Nicolet National Forest occur in

Table 1.—*Land ownership of forests in the western Lake States (from Powell* et al. *1994). Numbers indicate forest area in acres x 1000.*

Ownership type	Minnesota	Wisconsin	Michigan	Total
Public	(7,602)	(4,215)	(6,196)	(18,013)
Federal	2,019	1,429	2,438	5,886
State	3,078	580	3,571	7,229
County/Local	2,506	2,206	187	4,899
Private	7,171	10,706	11,225	29,102
Industrial	(751)	(1,179)	(1,981)	(3,911)
Total	14,773	14,921	17,442	47,136

Table 2.—*Relative frequency of common bird species in point surveys of national forests in the western Great Lakes region. Numbers indicate the percent frequency of each species relative to the frequency of the most abundant species (ovenbird). Numbers in parentheses indicate the rank abundance of each species in terms of frequency. Nicolet National Forest data were derived from randomly chosen roadside points sampled during 1993-1994. Data from the other national forests are from Niemi and colleagues. Neotropical migrants are shown in bold. Note that several important Neotropical migrants (e.g., black-throated blue warbler, olive-sided flycatcher) are well established in these forests but are not among the most common species listed here.*

Species	Relative Abundance (% frequency of most abundant species)			
	Nicolet NF (n=200)	Chequamegon NF (n=396)	Superior NF (n=522)	Chippewa NF (n=413)
Ovenbird	100 (1)	100 (1)	100 (1)	100 (1)
Red-eyed vireo	83 (2)	89 (2)	75 (4)	98 (2)
American robin	53 (3)	40 (7)	36 (12)	22 (24)
Chestnut-sided warbler	47 (4)	40 (8)	65 (5)	51 (6)
Rose-breasted grosbeak	46 (5)	29 (16)	34 (14)	20 (26)
Nashville warbler	44 (6)	58 (4)	89 (2)	68 (3)
White-throated sparrow	43 (7)	36 (9)	81 (3)	46 (8)
Mourning warbler	40 (8)	22 (19)	44 (9)	24 (23)
Black-capped chickadee	37 (9)	29 (15)	17 (27)	24 (23)
Blue jay	37 (10)	34 (11)	22 (22)	29 (17)
Hermit thrush	36 (11)	48 (5)	33 (17)	46 (9)
Black-throated green warbler	34 (12)	61 (3)	37 (11)	27 (19)
American crow	31 (13)	5 (51)	2 (62)	8 (43)
Least flycatcher	28 (14)	45 (6)	34 (14)	50 (7)
Indigo bunting	25 (15)	6 (45)	2 (64)	4 (53)
Song sparrow	23 (16)	13 (27)	14 (31)	31 (12)
Chipping sparrow	23 (17)	8 (40)	17 (28)	29 (17)
Veery	20 (18)	36 (10)	61 (6)	60 (4)
Cedar waxwing	19 (19)	3 (65)	8 (39)	13 (34)
Eastern wood-pewee	19 (20)	33 (13)	10 (33)	35 (11)
Winter wren	18 (21)	30 (14)	44 (8)	29 (15)
Scarlet tanager	16 (22)	20 (20)	10 (34)	30 (14)
Black-and-white warbler	15 (23)	33 (13)	46 (7)	20 (25)
Common raven	14 (24)	5 (51)	2 (58)	2 (66)
Common yellowthroat	14 (25)	23 (18)	21 (25)	51 (5)
Yellow-rumped warbler	14 (26)	28 (17)	32 (18)	30 (14)
American redstart	14 (27)	12 (30)	27 (19)	37 (10)
Golden-winged warbler	14 (28)	4 (55)	3 (54)	12 (36)
Great crested flycatcher	13 (29)	16 (24)	2 (62)	18 (27)
Yellow-bellied sapsucker	12 (30)	19 (23)	21 (25)	25 (21)
Blackburnian warbler	9 (34)	19 (21)	43 (10)	25 (21)
Brown creeper	9 (42)	19 (23)	11 (32)	13 (33)
Red-breasted nuthatch	12 (31)	15 (25)	19 (26)	15 (31)
Brown-headed cowbird	9 (35)	14 (26)	0 (85)	16 (29)
Yellow-bellied flycatcher	1 (85)	13 (28)	27 (20)	12 (37)
Canada warbler	1 (99)	12 (29)	34 (15)	5 (51)
Northern parula	6 (44)	11 (32)	23 (21)	17 (28)
Pine warbler	4 (51)	11 (33)	4 (50)	29 (17)
White-breasted nuthatch	9 (35)	10 (35)	3 (54)	15 (30)
Golden-crowned kinglet	2 (67)	5 (51)	14 (30)	11 (38)
Magnolia warbler	5 (48)	3 (62)	34 (16)	5 (51)
Swainson's thrush	2 (75)	2 (67)	16 (21)	2 (63)
Tennessee warbler	0 (119)	1 (81)	21 (23)	1 (77)

more that 10 percent of the points sampled. Patterns of relative abundance, based on frequency of occurrence among sample points, are similar among the four national forests; the 10th most common species, for example, are 37 percent, 36 percent, 33 percent, and 37 percent as frequent as the most abundant species in the Nicolet, Chequamegon, Superior, and Chippewa National Forests, respectively (table 2)

Virtually all bird species of these forests can be found in other parts of North America. Only five (Nashville warbler, black-throated blue warbler, blackburnian warbler, chestnut-sided warbler, and Canada warbler) have the centers of their geographic range and abundance within the region of mixed northern forest Udvardy (1963). Results from the North American Breeding Bird Survey (Price *et al.* 1995), however, show that the western Lake States support some of the continent's richest assemblages of bird species, particularly long-distance neotropical migrants, flycatchers, thrushes, and warblers. Several bird species not mentioned by Udvardy also seem to have centers of abundance or at least regional centers of abundance in the western Lake States and nearby parts of Canada, including broad-winged hawk, ruffed grouse, black-billed cuckoo, yellow-bellied sapsucker, pileated woodpecker, yellow-bellied flyctcher, least flycatcher, brown creeper, sedge wren, golden-crowned kinglet, veery, golden-winged warbler, magnolia warbler, northern parula, Cape May warbler, black-throated green warbler, pine warbler, black-and-white warbler, ovenbird, mourning warbler, Connecticut

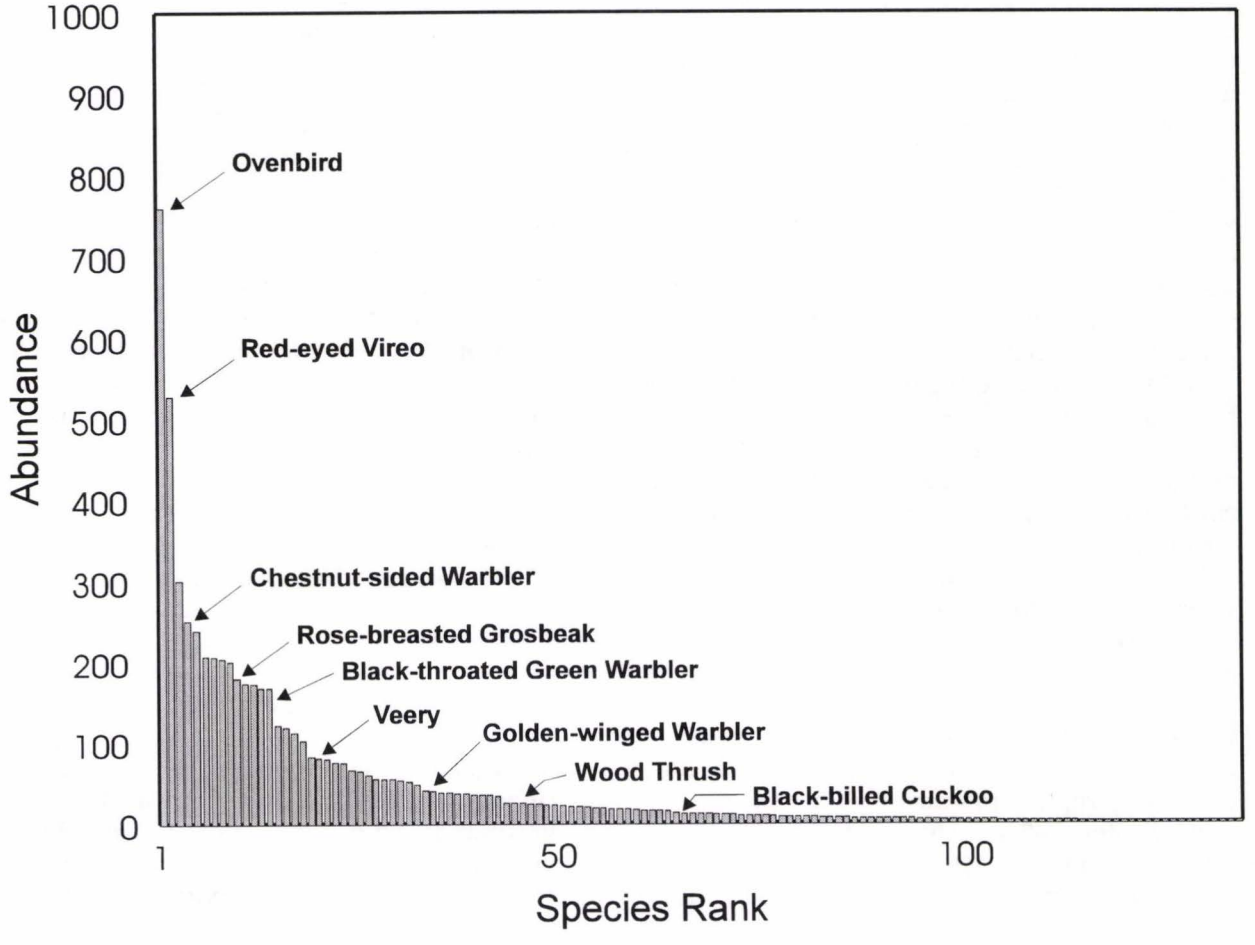

Figure 2.—*Rank abundance of bird species at 451 roadside point counts sampled during the Nicolet National Forest Bird Survey between 1992 and 1995. Abundance is the total number of individuals seen or heard during all 451 counts combined.*

warbler, rose-breasted grosbeak, purple finch, pine siskin, and evening grosbeak (Price *et al.* 1995). Kirtland's warbler, of course, breeds only in this region. Temple *et al.* (1979) concluded that the rich diversity of birds in the western Lake States is due to the co-occurrences of northern and southern bird species. Other explanations include the widespread mix of conifer and deciduous forests (related to Temple *et al.*'s conclusion), the complex glacial topography and associated wetlands, the large extent of forest habitat, and perhaps biophysical effects associated with the Great Lakes.

Population Trends

Except for the passenger pigeon, a species probably at the edge of its range in the western Lake States (Schorger 1955), no bird species of this region has gone extinct during historic times, and only one breeding neotropical migrant (Kirtland's warbler) is considered to be globally endangered or threatened. On a continental scale, however, several NTMBs characteristic of Lake States forests have declined significantly since the 1960's (Peterjohn and Sauer 1994, Price *et al.* 1995), including olive-sided flycatcher, eastern wood-pewee, wood thrush, veery, golden-winged warbler, and indigo bunting. Others, such as least flycatcher, Tennessee warbler, Nashville warbler, Cape May warbler, mourning warbler, Canada warbler, and rose-breasted grosbeak, have declined significantly during the past decade. Decreasing regional trends also have been documented for least flycatcher, golden-winged warbler, blackburnian warbler, and northern parula (Hunter *et al.*1995). Significant continental increases have been reported for hermit thrush (a short-distance migrant), solitary vireo, Philadelphia vireo, red-eyed vireo, blackburnian warbler, yellow-rumped warbler, ovenbird, and pine warbler (Peterjohn and Sauer 1994, Price *et al.* 1995).

Blake *et al.* (1994) described significant annual changes in abundance for approximately half of the neotropical migrant bird species in northwestern Wisconsin and Upper Michigan between 1986 and 1992. Declines followed severe drought during 1987 and 1988, but by 1992, most species had recovered to pre-drought population levels. Wood thrush, mourning warbler, and chestnut-sided warbler continued to show lower population levels, and golden-winged warbler declined sharply after an increase between 1986-1989. Trends in Wisconsin and Michigan were significantly different for 5 of 16 neotropical migrants (red-eyed vireo, Nashville warbler, northern parula, ovenbird, and rose-breasted grosbeak). The complexities of population changes noted by Blake *et al.* and other investigators (e.g., Sauer and Droege 1992, Villard and Maurer 1995, James *et al.* 1996) suggest that trends in neotropical migrant bird populations are typically not uniform in time and space.

Of 47 bird species for which the Midwest is of particular importance (Probst and Thompson 1996), 36 are present in the western Lake States, including 9 grassland species, 9 deciduous and 8 coniferous forest species, and 13 species of wet or dry shrublands. (Several species are listed in more than one category). Most of the "shrublands" represent early stages of forest succession. Early seral stages in the Lake States have particular importance for sedge wren, golden-winged warbler, chestnut-sided warbler, and mourning warbler. The region provides exclusive habitat (early successional jack pine) for Kirtland's warbler (*Dendroica kirtlandii*). Northern forests appear to be important centers of abundance for least flycatcher, veery, black-throated blue warbler, Nashville warbler, blackburnian warbler, Canada warbler, and rose-breasted grosbeak (Price *et al.* 1995). The region probably provides valuable source habitat (Brawn and Robinson 1996) for broadly distributed species like wood thrush, ovenbird and black-throated green warbler. Many of the bird species found in wet shrub, lowland conifer, and upland mixed forests (e.g., Cape May warbler, northern parula, northern waterthrush) have centers of abundance in neighboring parts of Canada, while others characteristic of deciduous forest types (e.g., Cerulean warbler, scarlet tanager) have centers of their ranges to the southeast.

Geographic Patterns

Because the majority of species are rare or uncommon (fig. 2), gaps occur in the geographic distribution of many forest bird species in the western Lake States. Even a relatively common species like least flycatcher shows a localized and somewhat unpredictable distribution pattern (MacQueen 1950, Montgomery 1995). In the Nicolet National Forest, for example, this species exhibits a strong preference for northern hardwoods (table 3),

yet individuals are present at only about 50 percent of all point counts in mature or immature northern hardwood forests. Individuals are observed consistently at an unexpectedly high number of sites, and are consistently absent from an unexpectedly high number of other sites (fig. 3). Davis (1996) used Monte Carlo randomization tests to show that 34 of the 65 most common forest or woodland bird species in the Nicolet National Forest show aggregations at some scale, even taking into account the spatial distribution of forest types. Non-random or patchy distributions imply that factors such as geographic gradients, fine scale habitat characteristics, landscape context, local catastrophes, and perhaps socially mediated "centers of abundance" significantly affect the distribution of breeding birds in northern forest landscapes.

At least 11 species of neotropical migrants show statistically significant differences in frequency between the southern and northern portions of the Nicolet National Forest, a maximum distance of only 160 km (Howe *et al.* 1995, Davis 1996). These differences are not

Table 3.—*Habitat associations of forest birds in the Nicolet National Forest based on results of 736 point counts between 1989-1994. Results indicate the number of counts where the species was recorded (n) and the Chi² value from a contingency table based on the assumption that birds should occur in proportion to the total number of counts in each of 10 forest categories. Nine species which occurred in >30 counts (black-billed cuckoo, common flicker, common raven, downy woodpecker, great crested flycatcher, hairy woodpecker, pileated woodpecker, white-breasted nuthatch, and wood thrush) are not included because they showed no significant forest habitat associations (Chi² < 16.92, P>.05). American robin (n=228) was not included even though its Chi² value was marginally significant (Chi² = 17.17).*

Species	n	Chi²	Preferred Forest Type(s)	Avoided Forest Type(s)
American crow	76	29.14	Aspen/Birch, Balsam fir/Spruce	Hemlock, Mature hardwoods
Black-and-white warbler	101	57.48	Lowland conifers, Lowland hardwoods	Immature hardwoods
Blackburnian warbler	92	83.33	Hemlock, White pine	Imm. hardwoods, Aspen/Birch
Black-capped chickadee	188	43.54	Balsam fir/Spruce	Imm. hardwoods
Black-throated blue warbler	41	33.73	Imm. hardwoods, Mat. hardwoods	Balsam fir/Spruce, Hemlock
Black-throated green warbler	256	172.41	Mature hardwoods, Hemlock	Lowland conifers
Blue jay	232	43.30	Red pine, Balsam fir/Spruce	Mature hardwoods
Brown creeper	69	31.86	Lowland hardwoods	Aspen/Birch
Brown-headed cowbird	36	20.08	Jack pine	Lowland conifers
Canada warbler	22	53.79	Lowland hardwoods, Balsam fir/Spruce	Mat./Imm. hardwoods
Cedar waxwing	91	45.78	Lowland conifers, Red pine	Mat. hardwoods
Chestnut-sided warbler	144	92.51	Aspen/Birch	Hemlock, Imm./Mat. hardwoods
Chipping sparrow	57	114.54	Red pine, Balsam fir/Spruce, Jack pine	Imm./Mat. hardwoods
Eastern wood pewee	146	99.77	Mat. hardwoods, Imm. hardwoods	Lowland conifers
Golden-crowned kinglet	52	59.69	Lowland conifers, Balsam fir/Spruce	Aspen/Birch
Golden-winged warbler	30	29.21	Lowland conifers, Aspen/Birch	Mat./Imm. hardwoods
Hermit thrush	267	32.31	Jack pine, Lowland conifers	Mature hardwoods
Indigo bunting	66	40.97	Aspen/Birch	Mature hardwoods
Least flycatcher	170	138.28	Mat. hardwoods, Imm. hardwoods	Lowland conifers, Balsam fir/Spruce
Mourning warbler	122	31.25	Aspen/Birch	Hemlock, Jack pine
Nashville warbler	212	234.99	Lowland conifers	Imm. hardwoods, Mat. hardwoods
Northern parula	72	39.75	Lowland conifers	Aspen/Birch, Red pine
Ovenbird	559	222.37	Imm. /Mat. hardwoods, Aspen/Birch	Lowland conifers
Pine warbler	28	98.08	White pine, Red pine	Imm. hardwoods, Mat. hardwoods
Red-breasted nuthatch	89	39.38	Balsam fir/Spruce, Hemlock, Red pine	Immature hardwoods
Red-eyed vireo	477	197.89	Mat./Imm. hardwoods, Aspen/Birch	Lowland conifers, Red pine
Rose-breasted grosbeak	262	93.49	Mat./Imm. hardwoods, Aspen/Birch	Lowland conifers, Jack pine
Scarlet tanager	133	20.67	Balsam fir/Spruce, Mat. hardwoods	Lowland conifers, Jack pine
Solitary vireo	55	25.92	Lowland conifers, Hemlock	Mat. hardwoods, Aspen/Birch
Veery	131	18.52	Imm. hardwoods, Lowland hardwoods	Red pine
White-throated sparrow	237	192.68	Lowland conifers, Balsam fir/Spruce	Imm. hardwoods, Mat. hardwoods
Winter wren	144	82.33	Lowl. conifers, Lowl. hwds., Hemlock	Imm. hardwoods, Aspen/Birch
Yellow-bellied flycatcher	41	113.97	Lowland conifers	Mat./Imm. hardwoods, Aspen/Birch
Yellow-bellied sapsucker	83	41.86	Lowland hardwoods, Imm. hardwoods	Red pine, White pine
Yellow-rumped warbler	92	72.48	Lowland conifers, Red pine, Jack pine	Mature hardwoods, Imm. hardwoods

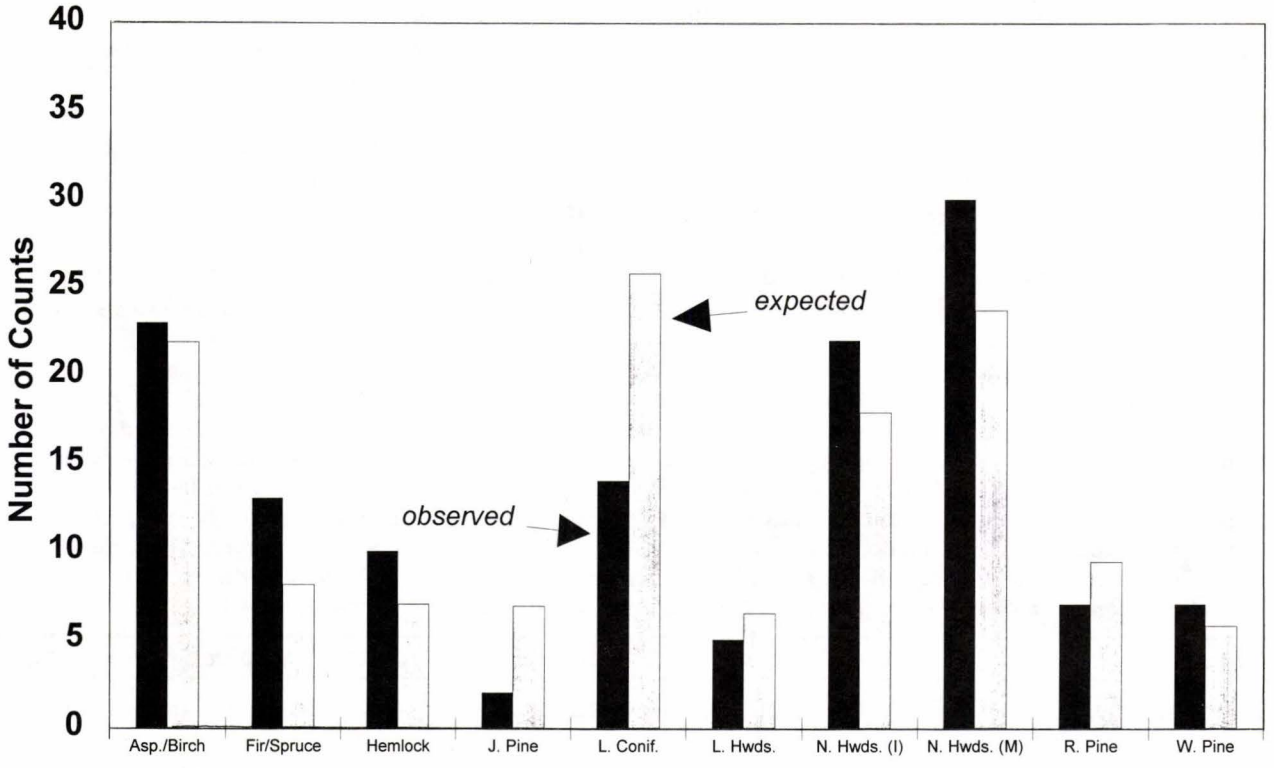

Figure 3.—*Distribution of scarlet tanagers among different forest types in the Nicolet National Forest based on habitat-based point counts between 1989-1994. Number of counts is the total number of point counts in a given habitat where scarlet tanagers were observed. Forest types are defined by dominant tree species: Asp./Birch = aspen/birch, J. Pine = jack pine, L. Conif. = lowland conifers, L. Hwds. = lowland hardwoods, N. Hwds. (I) = immature northern hardwoods (dominated by trees 31-70 years old), N. Hwds. (M) = mature northen hardwoods (dominated by trees > 70 yrs. old), R. Pine = red pine, and W. Pine = white pine. Expected numbers of counts were calculated by assuming that scarlet tanagers occur randomly among sites, so that the probability of an individual being detected in a given forest type is directly proportional to the number of counts in that type.*

due to the absence of favored habitat types, although the overall extent of many habitat types varies from south to north, as do the centers of distribution of the bird species. Yellow-billed cuckoo, great crested flycatcher, northern oriole, indigo bunting, scarlet tanager, and wood thrush are more common in southern half of the Forest, while olive-sided flycatcher, least flycatcher, black-throated blue warbler, chestnut-sided warbler, and black-throated green warbler are more common in the northern half. Results from the Michigan Breeding Bird Atlas (Brewer *et al.*, 1991) show both an extension and a parallel to these geographic trends over a broader north-south gradient.

Bird / Ecosystem Relationships at Landscape and Local Scales

Landscape relationships

On a larger scale, landscape variables have been shown to be correlated with bird abundance in the northern forest region. Hawrot and Niemi (1995) found that breeding bird species were influenced by the extent of different habitat types and the local abundance of forest edge. Pearson (1994) found that habitat types surrounding a forest stand influenced the relative abundance of some bird species within the stand. For instance, if a mature aspen stand is surrounded by coniferous forest, then bird species associated with

coniferous trees are more likely to be found within the stand, even if the predominant trees are deciduous. A similar result was found by Elias (1996). McRae (1995), working with data from the Nicolet National Forest Bird Survey and high altitude photographs, showed that the local abundances of 10 neotropical migrants (least flycatcher, wood thrush, Swainson's thrush, red-eyed vireo, solitary vireo, black-throated green warbler, blackburnian warbler, black-throated blue warbler, black-and-white warbler, and scarlet tanager) are significantly correlated with landscape variables such as the percent upland conifer-hardwood forest within 500 m (positively) and the percent upland open habitat within 250 m (negatively). These studies are important because they show that bird species composition and abundance are influenced by the configuration of forest vegetation *surrounding* a local site, thus explaining much of the local variability discussed below. They also demonstrate the utility of landscape-level tools such as high altitude images and geographic information systems in evaluating bird habitat quality.

Succession

Several studies in Laurentian mixed forests have evaluated the relationship between breeding bird abundances and forest succession. Webb *et al.* (1977), Steffen (1984), Morgan and Freedman (1986), Probst *et al.* (1992), and Helle and Niemi (1996) examined bird species composition in stands of different ages after logging. Probst *et al.* (1992) noted an almost complete turnover in bird species during succession from clearcuts to older aspen forests in Minnesota and Michigan. Bird species are added during early succession as vegetation life forms (e.g., shrubs, tree canopy, bark surface) become more prominent parts of the forest ecosystem. Steffen's (1984) comparison of young vs. older aspen stands is generally consistent with the results of Probst *et al.* Helle and Niemi (1996) found that bird species diversity increased during succession up to the mature forest stages, then declined slightly in the oldest forests. Neotropical migrants of early successional stands include black-billed cuckoo, gray catbird, golden-winged warbler, common yellowthroat, Nashville warbler, chestnut-sided warbler, mourning warbler, and indigo bunting. Important neotropical migrants in mature northern forests include broad-winged hawk, least flycatcher, eastern wood-pewee, great crested flycatcher, wood thrush, red-eyed vireo, yellow-throated vireo, black-throated green warbler, blackburnian warbler, ovenbird, and scarlet tanager. Veery, rose-breasted grosbeak, American redstart, black-and-white warbler, black-throated blue warbler, and Canada warbler reach highest densities in intermediate-aged forests, but these species also occur in mature forests where canopy gaps modify the forest composition and structure.

Forest types

Beyond the coarse preference of species for general vegetation structure, bird-habitat relationships at the local level are complex and poorly understood. The ranges of most birds in the western Great Lakes forests are broad and encompass variations in habitat. Scarlet tanagers, for example, occur in nearly every forest habitat type in northern Wisconsin (fig. 4), even though they tend to prefer upland hardwood forests and avoid lowland conifers.

If birds show no preference for forest types, then their frequency in point samples should be proportional to the number of samples representing each type. Expectations based on the hypothesis of no preference can be compared with observed frequencies by a simple Chi^2 test of independence (Conover 1980). Results for birds of the Nicolet National Forest show some interesting patterns (table 3). Six of nine species which show no significant associations with forest type are cavity nesters. Apparently within-stand features (undoubtedly including the presence of suitable cavity trees) are more important than dominant tree species. Of the 35 species which were common enough to show habitat associations, 12 prefer lowland conifers, while only 10 species prefer upland hardwoods (mature or immature), the most widespread habitat type in the Nicolet National Forest. These simple tests ignore the effects of forest age, although regenerating stands (< 10 yrs old) were excluded from this analysis and only a few, small old-growth stands were available for sampling. Chi^2 values tend to be correlated (positively) with frequency of occurrence ($r = 0.66$, $p < 0.01$). Taking this relationship into account, strongest habitat associations are shown by Nashville warbler, white-throated sparrow, black-throated green warbler, least flycatcher, yellow-bellied flycatcher, and chipping sparrow (table 3). But conservation

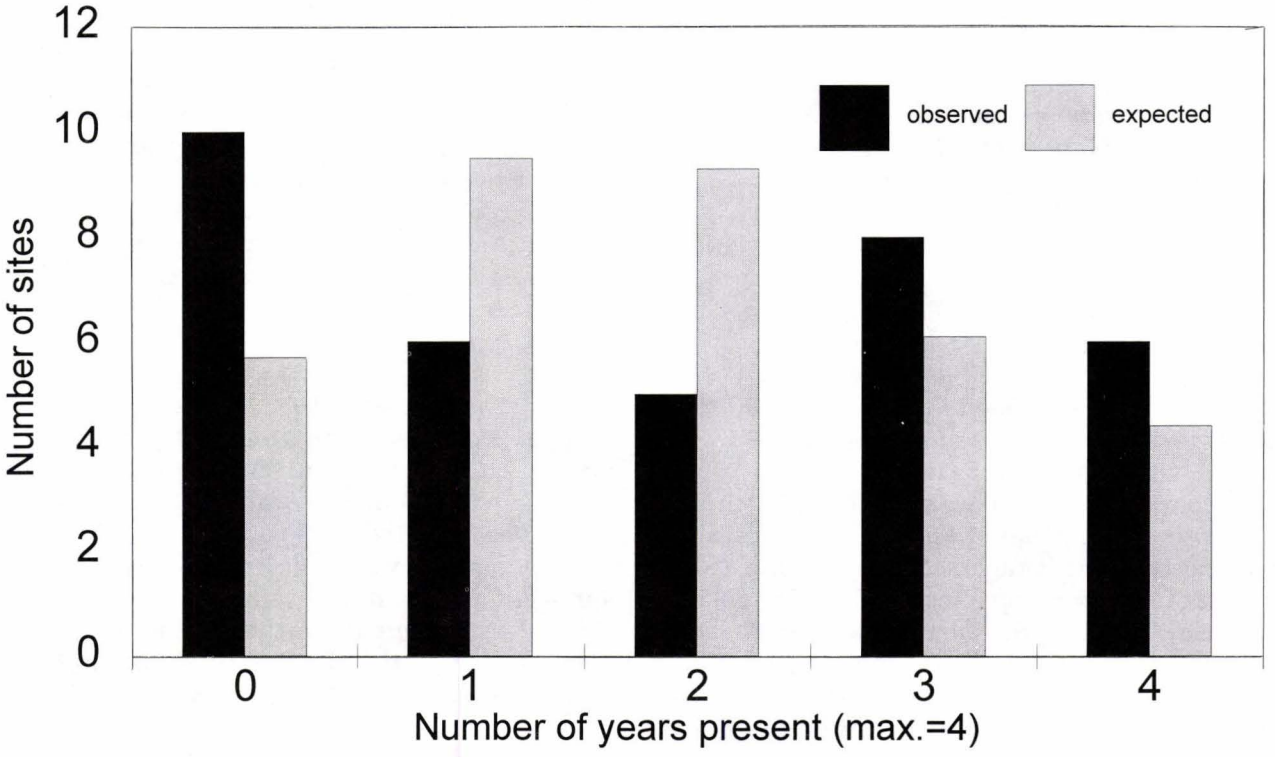

Figure 4.—*Fidelity of least flycatchers to 35 point localities dominated by northern hardwoods (> 31 yrs. old) in the southern Nicolet National Forest. Each point was sampled 4 times (bi-annually) between 1989 and 1995. Histograms show the number of sites where least flycatchers were observed during all 4 years, during all 3 years, etc. Expected numbers are means from 1,000 computer trials, where the bird occurrences were randomly allocated to the 35 sites over 4 "years." Note that the birds tended to occur more oftern than expected at certain sites, and less often than expected at others.*

biology concerns the rare or less common species, so there is a need to design step-wise assessments for them at appropriate scales.

Regional differences in bird-habitat associations are evident for a number of species. Black-throated green warblers in Door County, Wisconsin, for example, are abundant in lowland conifer forests (Zovnic and Howe 1995). In contrast, this species avoids lowland conifers in the Nicolet National Forest (table 3). Connecticut warblers are associated with jack pine barrens in northern Wisconsin (Robbins 1991), whereas they occur within bogs in Minnesota (Niemi and Hanowski 1992) and at bog edges and in aspen parklands in Michigan (Brewer *et al.* 1991). Thus, it is difficult to extrapolate detailed surveys about abundance or frequency from one subregion, landscape or ownership to other contexts without completing local bird surveys.

Habitat features

Dominant tree species do not necessarily define optimal or suboptimal habitats. MacArthur *et al.*'s (1962) classical study demonstrated that bird species diversity is positively correlated with the vertical diversity of foliage layers. Beals (1960) and Temple *et al.* (1978) showed that veery, black-throated blue warbler, and Canada warbler increase in abundance with increasing density of understory vegetation. Zovnic (1995) observed that increased reproductive success of ground nesting songbirds (ovenbird, hermit thrush, veery, and wood thrush) in the Nicolet National Forest is positively associated with the density of shrubs and composition of the ground layer (especially the abundance of *Lycopodium*). Pearson (1994) found that the abundance of veeries in north-central Minnesota is correlated negatively with the frequency of balsam

fir (which promotes a sparse understory) and percent canopy cover. Winter wrens favor sites with fallen logs or extensive woody debris (Wolf and Howe 1992).

FORESTRY AND BIRD POPULATIONS

Controversies have arisen because studies from other ecosystems have been applied to forestry impact assessments in the western Lake States. Kuhlman (1990) argues that forest harvesting has reduced populations of migratory birds in the Great Lakes region. Lorimer (1994), on the other hand, sees little evidence that conventional timber harvest in North America has been a significant factor in declines of neotropical migratory bird populations. The complex nature of population trends and the limited availability of baseline data make either claim speculative, but evidence from recent field studies provides some means for assessing the impacts of management activities on birds of the western Great Lakes landscape.

Habitat Destruction

Elimination of forest habitat obviously reduces local populations of forest-dependent bird species. Questions about conservation of NTMBs as well as other species must begin with at least a general estimate of species' population sizes. A sample of 200 points distributed approximately randomly in the Nicolet National Forest yielded 2,549 individual birds of 88 species (fig. 2), excluding aquatic species (e.g., common loon, great blue heron), strictly open country species (e.g., bobolink, barn swallow), and nocturnal species (e.g., barred owl, common nighthawk). Most of the detections were singing males; for the sake of this simple exercise, let us assume that they all represent resident, mated and monogamous males, leading to an estimate of 2,549 pairs (however, studies have shown that many males detected are transient). Studies by Wolf *et al.* (1995), have shown that most singing birds can be detected within 125 m. The 200 randomly sampled points therefore cover about 0.25 percent of the 389,000-ha Nicolet National Forest. Extrapolation from these samples (but see above) gives an estimate of 1,010,563 singing males representing more than 100 species (see Preston 1962). The entire western Lake States region described by Frelich (1995) is about 51 times larger than the Nicolet National Forest; therefore we estimate a regional population of at least 51.5 million singing males. Approximately 109 species of forest birds (excluding strictly open country species, etc.) are known to breed in this region. Based on our documented pattern of relative abundance (fig. 2), at best 15 to 20 species are likely to be represented by more than a million breeding pairs in the western Great Lakes forests, assuming that the Nicolet National Forest is representative of the region as a whole. (Of course, populations of most species also occur in Canada and elsewhere in the U.S.) At the other extreme, 24 of the 88 species in our sample were represented by fewer than 5 individuals; 10 species were encountered only once, and at least 20 potentially breeding species were not encountered at all. Again, using our assumption that the Nicolet National Forest is representative of the region, populations of these species in northern forests must be less than 100,000. This analysis is misleading if species that are rare in the Nicolet National Forest are common elsewhere, but our earlier findings (table 2) show that species which are common in the Nicolet National Forest also are common in other national forests of the western Great Lakes region and vice versa. Woodland or forest edge birds (e.g., brown thrasher, yellow-billed cuckoo, rufous-sided towhee) clearly are better represented in other less extensively forested regions, but we are still left with a significant number of forest or forest/wetland birds (table 4) which occur in very low numbers. Several of these species (yellow-bellied flycatcher, Canada warbler, magnolia warbler, Tennessee warbler) are at least locally common in more northern regions; even in the Superior National Forest, however, only Canada warbler is among the 15 most abundant species.

Destruction or degradation of habitats occupied by these and other uncommon species obviously can have serious consequences for their long-term survival. In these cases, specific distribution patterns must be identified, and specific breeding localities must be documented in order to avoid inadvertent loss of critical areas. Clearly, multi-scale studies need to be designed to assess the distribution and dynamics of less common species. These species may be highly selective or variable in their distribution.

Table 4.—*Northern forest NTMB's that are rare or uncommon across most of their geographic range. Species are included in this list if they are known to breed in the western Lake States and fewer than 400 (of >3000) Breeding Bird Survey routes (= # BBS Routes) are used to calculate their North American population trends between 1966 and 1993 (from Price* et al. *1995).*

Species	# BBS Routes
Yellow-bellied flycatcher	243
Philadelphia vireo	189
Golden-winged warbler	341
Tennessee warbler	332
Cape May warbler	242
Palm warbler	81
Bay-breasted warbler	215
Cerulean warbler	326
Connecticut warbler	109

Habitat Degradation

Land use by humans can affect the *quality* as well as the quantity of habitats, even for relatively widespread bird species. Howe *et al.* (1991) and others (Donovan *et al.* 1996) have shown that even local habitat degradation can cause extinction of widespread populations if significant density independent dispersal occurs from high quality habitats. Old growth forests dominated by trees > 100 years in age have been replaced extensively during the past century by younger forests (Frelich 1995). Hence, we might expect that species characteristic of old growth have declined and perhaps are now vulnerable to extinction. Howe and Mossman (1996) have compared birds of the Sylvania Wilderness in northern Michigan with birds of nearby managed forests. In general, differences in species composition are quantitative rather than qualitative. Red-shouldered hawk, and chimney swift, for example, are significantly more abundant in old growth than they are in managed forests, but no bird species is today restricted to old-growth forests in the western Lake States. Of course, presence of a bird in managed forest does not insure that the habitat can sustain a population (Van Horne 1983, Donovan *et al.* 1996), so a multi-scale approach to assessment is essential (Freemark *et al.* 1993).

Howe and Mossman (1996) found that more intensively managed (even-aged) hardwood forests support significantly lower species richness and abundances of neotropical migrants than do less intensively managed (uneven-aged) forests. Several species characteristic of forest gaps and understory (e.g., veery, black-throated blue warbler) show higher abundances in uneven-aged managed forests than in old growth. These results parallel findings from other regions (Thompson *et al.* 1992, Welsh and Healy 1993).

Evidence from our recent field studies suggests that the selective elimination of hemlock and other conifers from Lake States forests affects several neotropical migratory species negatively (also see Benzinger 1995). Blackburnian warbler, black-throated green warbler, pine warbler, and solitary vireo are locally common in the region as a whole, but populations are generally absent or less abundant in sites with few large conifers (Howe and Mossman 1996).

Certain neotropical migrants clearly have increased as a result of forestry practices. Chestnut-sided warbler, mourning warbler, magnolia warbler, golden-winged warbler, Canada warbler, and Nashville warbler were uncommon during pre-settlement times (Morse 1989), but populations increased as older forests were replaced by forests of aspen and young hardwoods. More recently, several of these species have decreased again as secondary forests have matured (Peterjohn and Sauer 1994). Golden-winged warblers have expanded from shrub swamps and open tamarack swamps to clearcuts and other young seral stages, but their populations today are declining. Canada warbler, Magnolia warbler, and black-throated blue warbler are usually associated with "gaps" in Lake States forests caused by natural disturbances or selective logging.

Effects of forest management practices on neotropical migrants deserve further study. Even in the cases described above, where the consequences of land use on neotropical migratory bird populations are somewhat predictable, the effects are species-specific. In short, no single recipe of land management affects all species in the same way, so broad-scale planning is necessary to accommodate all species and other objectives of land management (Thompson *et al.* 1993). Evidence has shown that many species are capable of using alternative habitats if their original

habitats are modified, but our inadequate knowledge about these habitat responses justifies long-term studies and population monitoring.

Landscape Modification and Biogeography

Area and isolation

In addition to its effects on the quantity and quality of available habitat, forest management also affects the configuration of habitats (Freemark *et al.* 1995). The familiar relationship between species richness and habitat area (MacArthur and Wilson 1967, Lynch and Whigham 1984, Robbins *et al.* 1989, Johns 1993) has been documented for birds of true islands in the Lake Superior region (Beals 1960, Rusterholz and Howe 1979) and for birds of forest habitat islands in Ontario (Freemark and Merriam 1985, Villiard *et al.* 1995) and elsewhere. Isolation effects in northeastern North American forests have been documented by Fritz (1979) and Benkman (1994). Area and isolation effects probably also apply to birds in discrete habitats like conifer bogs, sedge meadows, and even clear cuts, but these biogeographic patterns are more obscure when the matrix between optimal patches is habitable. Old-growth forest remnants, for example, are both small in area and isolated from other such remnants in the western Lake States (Frelich and Lorimer 1992). Most birds of these old-growth forests, however, also occur in managed forests. The complex juxtaposition of different habitat types complicates our understanding of population dynamics of northern forest birds (see Donovan *et al.* 1996). Future investigations along the lines of those by McRae (1995), Hawrot and Niemi (1996), and Elias (1996) are likely to be fruitful in understanding the biogeography of neotropical migrants in the western Great Lakes forests. Studies by Probst and Hayes (1986), Donovan *et al.* (1995) and Zovnic (1995) suggest that source-sink dynamics might be important for both rare and common species in this region, although demographic data are available for only a few species. Given recent emphasis on large-scale management policies (Kessler *et al.* 1992), additional research on geographic distributions and spatial dynamics of NTMB populations will help provide more specific guidance for decision makers.

Edge effects

Besides affecting the configuration of feeding and nesting resources, changes in the forest landscape can significantly influence species interactions. Indeed, large-scale changes in species interactions might comprise a greater and more insidious threat to breeding bird populations than reduction of habitat area. Perhaps the best-studied (and most controversial) effect of landscape changes on species interactions is the relationship between forest edge and nest predators or brood parasites.

Historically, edges were thought to be good habitats for wildlife. Edges represent the places where two distinctly different habitats merge. Often they are areas of relatively high productivity (Angelstam 1992), but many recent studies have shown that edges also are associated with increased incidence of nest predation and nest parasitism by brown-headed cowbirds (e.g., Wilcove 1985, Brittingham and Temple 1983, Robinson *et al.* 1995). Most of these studies, however, have been conducted at distinct edges (often called *hard edges*) such as those between forests and agricultural fields. Edges in the western Great Lakes Forests are more likely to be transitory or *soft edges*, occurring between qualitatively different forest or woodland habitat types.

Detailed studies of nest predation and nest parasitism within a predominately forested landscape are uncommon. Robinson *et al.* (1995) studied nest predation and nest parasitism rates in a variety of forested areas in the midwestern United States. They found that both nest predation and nest parasitism rates were negatively related to the percentage of forest cover in the landscape. Nest predation and parasitism rates were highest in areas with the most open agricultural or urban land area and lowest in areas with the most forest area. Robinson *et al.* (1995) showed that in certain areas predators destroyed as many as 80 percent of the nests for some bird species, but it is not clear if high predation is the result of the landscape composition or edge effects. Ambuel and Temple (1983) report that the negative effects of edges may penetrate 200 m or more into the forest interior.

The effects of nest parasitism by brown-headed cowbirds are known to be concentrated near habitat edges. The brown-headed cowbird evolved in the western United States,

primarily in open country where it was closely associated with large mammals such as the American bison (*Bison bison*). As land to the east was cleared for agriculture and livestock, the brown-headed cowbird expanded to the east and north (Mayfield 1965). Female brown-headed cowbirds can move relatively long distances, sometimes 7.5 km or more in order to find suitable host species (Rothstein *et al.* 1984, Thompson 1994). Forest birds such as flycatchers, thrushes, vireos, warblers, and tanagers have became regular victims of brown-headed cowbirds in today's agricultural/forest landscape. Currently more than 100 bird species are known to be host. Cowbirds are relatively common in the western Great Lakes region, but they are still relatively uncommon or even rare in areas that are predominantly forested (e.g., northeastern Minnesota, parts of northern Wisconsin, and Upper Michigan). Even in these areas, however, the species can be locally common in forest areas that are fragmented by agricultural or urban environments, especially if livestock or horses are present.

Bird densities in predominantly forested landscapes often are higher near forest edges (e.g. Gates and Gysel 1978; Hanowski *et al.* 1993, 1995, 1996), but predation and nest parasitism rates in general are not well documented for extensively forested regions (Paton 1994). Those studies that have been published offer little consensus about the effects of edges on forest bird populations. Some studies have shown that forest fragmentation increases nest predation and reduces breeding success (e.g., Wilcove 1985, Small and Hunter 1988). However, increased predation rates associated with edges have not been found in all studies (e.g., Ratti and Reese 1988, Rudnicky and Hunter 1993, Small and Hunter 1988). Two studies on nest productivity within a logged forest mosaic in north central Minnesota showed mixed results. Fenske's (1995) study of more than 500 artificial nests found that predation rates were higher near "soft" edges between medium-aged (3-19 yr) and mature (>40 yr) aspen forests than near "hard" edges between recently logged (2-4 yr old) and mature aspen forests. Fenske (1995) also found that predation rates tended to decrease toward the forest interior at distances of 50 m and 100 m from both the soft and hard edges. Moreover, using a camera with a motion detector he recorded 28 nest predation events. The most common predator (8 of 28 events) was the fisher (*Martes pennanti*), with seven other mammal species recorded as predators.

In the same area, Hanski *et al.* (ms) found 311 natural nests of 33 bird species. They identified precise locations for each nest using a geopositioning system and calculated the nest success rates in relation to various edges. No relationships were found with distance to edges, but predation rates were highest for species nesting within the forest (62 percent) compared with birds nesting in open (42 percent) or regenerating (47 percent) habitat types. Brown-headed cowbird parasitism rates were relatively low (10 percent) and unsuccessful (2 percent). These studies emphasize that context of region, landscape, ecosystem type and age, and species associations must be considered before extrapolating local nest monitoring results.

Nest predation and parasitism are primary factors affecting reproductive success in forest birds (Martin 1992). These factors need to be studied with multi-scale approaches and well-defined hypotheses in varying forest landscapes. We can expect to find considerable variation in rates of reproductive success among different forest environments because of the underlying variation in regional populations and relative species abundances, the relative abundances and mix of predators, the abundance of brown-headed cowbirds, and the mix of forest types and ages across the landscape.

Ecological Imbalances and Species Interactions

The expansion of brown-headed cowbirds into the Lake States represents one of many ecological changes that affect the quality of Laurentian forest landscapes for NTMBs. Suppression of spruce budworm outbreaks and selective elimination of balsam fir negatively affect Cape May warbler and other species which specialize on local budworm outbreaks (Morse 1986). Invasions of gypsy moth populations into the Lake States (and perhaps the methods used to control them) have at least local effects on insectivorous birds (DeGraaf 1987). Increases in the abundance of crows due to high numbers of road-killed deer may lead to higher rates of nest predation on open-nesting songbirds. Recent increases and re-introductions of mammalian

predators like fisher and pine marten have led to increased nest predation rates, perhaps exacerbated by the absence of top-level carnivores like wolves and mountain lions in much of the region. Expansion of blue-winged warblers into the western Lake States affects the local abundance of golden-winged warblers through hybridization and competitive interactions (Murray and Gill 1978). Subtle (but important) ecological interactions like these are rarely addressed by biologists or resource managers because evidence about their severity is difficult to obtain. Despite our ignorance, complex species interactions are pervasive in nature and surely deserve less traditional approaches by researchers; their elusiveness and potential significance underscore the need for general surveys as a framework for continuous population monitoring (Freemark *et al.* 1993).

MANAGEMENT RECOMMENDATIONS

Our current understanding of bird populations in western Great Lakes forests provides several tools for NTMB conservation and management. Given the high diversity of bird species in this region, general planning cannot address the complex habitat associations and local distributions of all species. On the other hand, individual management plans for 50 or more species cannot be formulated at high levels of detail, let alone implemented, given today's limited conservation budgets. The most effective approach under these circumstances is to establish a clear problem orientation, leading eventually to species priorities. Thompson *et al.* (1993) used several criteria to rank the conservation priority of NTMBs in the Midwest but priorities may change rapidly. Data from specific management units (fig. 5) provide one means for identifying local species

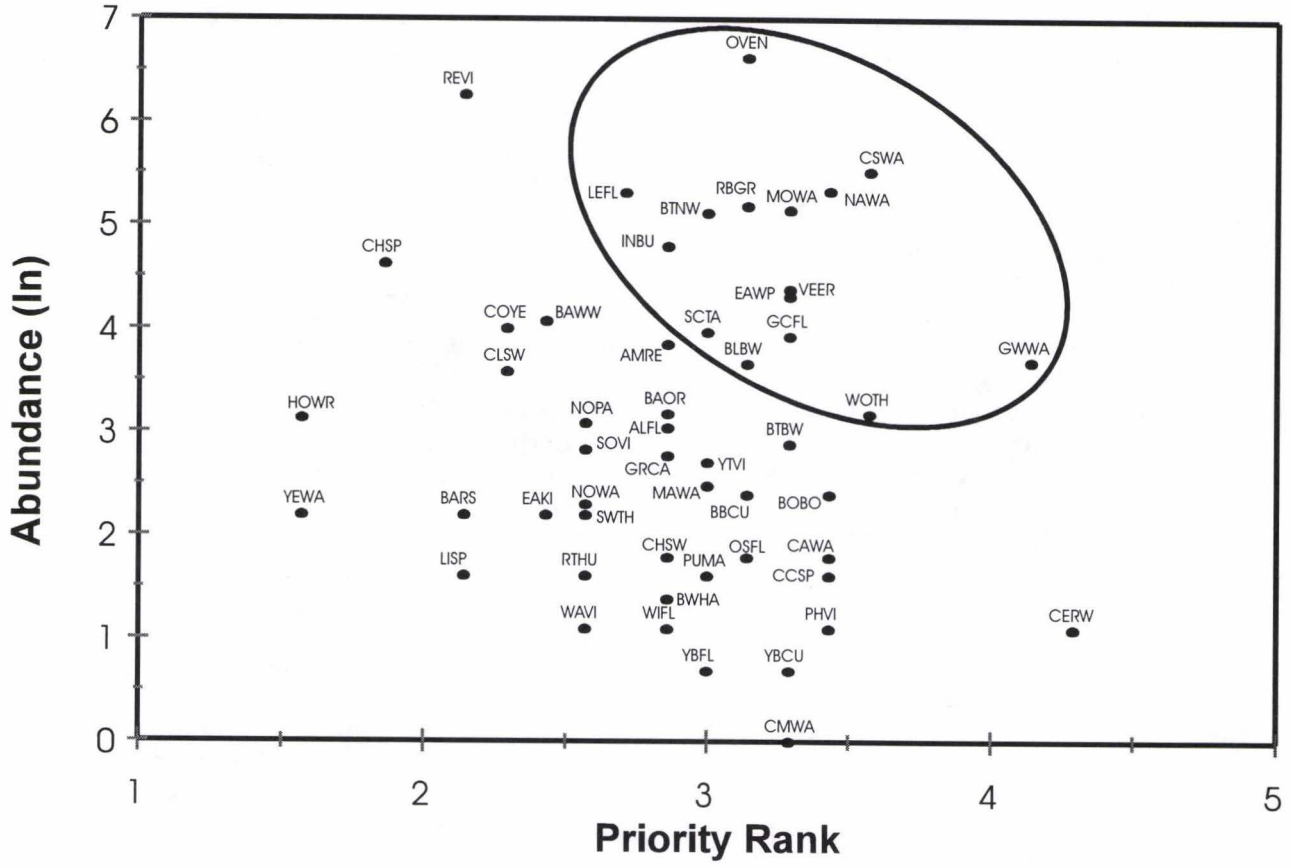

Figure 5.—*A suggested scheme for identifying conservation priorities of NTMB's in the Nicolet National Forest based on abundance in random roadside counts (1992-1995) and regional priorities recommended by Thompson* et al. *(1993). Priority species (circled) are both highly ranked by Thompson* et al. *(1993) and are relatively abundant in the Forest. Species abbreviations are standard American Ornithologists' Union codes formed by taking the first four letters of the bird's common name (if one word) or the first two letters of each word in its name (if two words), etc. Exceptions include the black-throated green warbler (=BTNW), blackburnian warbler (=BLBW), barn swallow (=BARS), and Cerulean warbler (CERW).*

priorities, which may differ from a continental perspective. If we accept the ranking scheme of Thompson *et al.*, (for the present) conservation actions will be most effective if aimed at species that are both highly ranked and abundant enough contribute to global populations. In other words, we look for species in the upper right of figure 5. In the Nicolet National Forest one species, golden-winged warbler, stands out. Other species with high priority include wood thrush, chestnut-sided warbler, Nashville warbler, mourning warbler, veery, rose-breasted grosbeak, great-crested flycatcher, eastern wood-pewee, and blackburnian warbler. Middle priority species with declining populations (e.g., olive-sided flycatcher) deserve special attention (Probst and Thompson 1996). Abundant species like ovenbird, black-throated green warbler, and least flycatcher also fare well in this analysis. If we use a ranking system from the northeastern U.S. and adjacent parts of Canada (Smith *et al.* 1993), or from the rest of the Midwest (Probst and Thompson 1996), black-throated blue warbler and Canada warbler also emerge as high priorities.

Of course, these priority ranks do not address globally rare species, year-round residents, and short distance migrants. Species like northern goshawk, yellow-bellied sapsucker, and brown creeper deserve attention in the western Great Lakes forests because their population densities are low or because the region encompasses a significant part of their geographic ranges. A separate system of ranking and population assessment (e.g., for raptors) will need to be developed for these species. Finally, NTMBs that are uncommon across their ranges (table 4) should be protected wherever they occur, especially since BBS data are less likely to detect population declines in these species.

We have emphasized the importance of specific information about the distribution and ecology of priority bird species. Even the best designed habitat protection plan will be ineffective if the target species is not present and breeding in the management area. Effective conservation plans must target specific areas within a much broader range where these species are known to occur and breed successfully (just as conservation efforts currently are aimed at bald eagle nests, etc.). Such information-intensive management is not possible for all organisms and issues, but the high public interest in birds and the foundation of information already available makes a top-down framework compatible with a empirically-based management program. Although local-level detail is important, scientists also must seek broad-scale patterns in order to implement step-wise, successive approximation of more integrated plans.

A systematic, standardized information base promotes cooperation among resource managers within and beyond the Lake States. We have documented significant geographic variations in abundance of many species, sometimes over distances of less than 200 km. Bird conservation measures will be most effective if they are aimed at areas of highest reproduction. Many bird species inhabiting this region's wet shrubs, lowland conifers, and upland mixed forests, for example, have a large portion of their geographic ranges in Canada, where conservation efforts might be more critical. Lessons from theory and practice, however, warn against ignoring peripheral populations (Takekawa and Beissinger 1989, Howe *et al.* 1991, Lesica and Allendorf 1995). Ultimately successful species protection will depend on informed, geographically diverse, and properly focused management plans.

High priority species occur in virtually all forest types and age categories (Probst and Thompson 1996, table 3). Indeed, several of the highest priority species in the Midwest as well as the western Great Lakes (fig. 5, table 3) occur in early successional forests. This can be interpreted as good news for advocates of intensive forestry, but the importance of mature forests also must be emphasized. Ongoing studies of birds in old-growth and managed forests (Howe and Mossman 1996) show that both neotropical migrants in general and priority species in particular are more productive in old-growth forests than in even-aged managed forests. This is not surprising given the overwhelming extent of old growth in pre-settlement landscapes (Frelich 1994). Thus, a regional perspective is needed for planning silvicultural systems across ecosystems and ownerhsips (Thompson *et al.* 1993).

Whereas existing information helps us establish species priorities and critical areas for conservation, our knowledge about ecological interactions in western Great Lakes forests is grossly inadequate. Land use activities influence area and edge effects, predator-prey

dynamics, source-sink interactions, and other landscape level processes; the specific consequences of forest management activities on these phenomena are very difficult to predict. The complex forest environment and the richness of bird species imply that meaningful predictions might never be possible with studies based on bottom-up interpretations of local populations.

However, ignorance about cumulative and indirect species interactions is not bliss for bird conservation. We suggest a simple approach for improving our knowledge about the ecological impacts of forest management. Each year, timber sales covering (overall) thousands of acres are administered by public agencies in the western Lake States. These sales are equivalent to experimental "treatments" of great significance for future planning efforts. If a sample of timber sales (i.e., "treatments") could be preceded by quantitative inventories of birds and their reproductive success, and followed by subsequent inventories using the same methods, the short-term, local effects of forest management activities could be tested directly. A data base of results, ideally shared across agencies and states, would yield a large and statistically powerful method for assessing the impacts of forest management. However, the method would be conceptually weak if not framed in a regional, multi-scale context (e.g. Thompson *et al.* 1992, Probst and Weinrich 1993). Continuous population monitoring of benchmark "controls," designed within a multi-scale framework of targeted surveys could help guide truly enlightened *adaptive management* (Walters 1979) of forest bird populations.

In summary, our recommendations for management of NTMB populations in western Great Lakes forests are modest: (1) outline realistic conservation goals at several administrative levels; (2) establish species priorities based on local field inventories and global or regional priority schemes (e.g., fig. 5); (3) identify where specific populations of priority species occur; (4) implement appropriate habitat management (or habitat preservation) measures at these specific localities, perhaps guided by ongoing experimental field studies; (5) coordinate planning strategies among forest management units and at multiple scales; and (6) design monitoring efforts strategically within a continental perspective. This multi-dimensional approach will improve chances that local efforts will have lasting global significance.

Answers to challenging questions about the impacts of regional population fluctuations, landscape configuration, area effects, harvest levels, edge effects, or altered species interactions on NTMB populations simply aren't available for the western Great Lakes forests, and extrapolation from studies in highly fragmented forest landscapes, or any local situation, are not necessarily valid. We have proposed a mechanism for obtaining answers to such questions by step-wise successive approximation (Freemark *et al.* 1993, Probst and Thompson 1996) based on continuous population monitoring. Although results and cumulative effects from such studies might not be evident for many years, these results provide tests of broader-scale assessments (Probst and Thompson 1996, Thompson *et al.* 1993) for long term forest management and bird conservation. Some of the most innovative management policies today might be those which generate answers for tomorrow's conservation problems.

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THE IMPORTANCE OF FLOODPLAIN FORESTS IN THE CONSERVATION AND MANAGEMENT OF NEOTROPICAL MIGRATORY BIRDS IN THE MIDWEST

Melinda G. Knutson[1], Jeffrey P. Hoover[2], and Erwin E. Klaas[3]

ABSTRACT.— Floodplain forests of the Central Forest Region of the Upper Midwest are found primarily on the floodplains of large rivers and include at least 12 forest cover types (species groups). Birds breeding in floodplain forests are affected by extensive variation in latitude, climate, hydrology, forest succession, and change caused by anthropogenic disturbances. New information from two recent studies, one on the Upper Mississippi River and one on the Cache River in southern Illinois, indicates that floodplain forest bird communities differ in species composition and in relative abundance from adjacent upland habitats. High abundances of some species are found in the floodplain and species, such as the prothonotary warbler, brown creeper, yellow-billed cuckoo, yellow-bellied sapsucker, and great crested flycatcher, show a clear preference for floodplain forests. A few species, such as the red-shouldered hawk and prothonotary warbler are dependent on these forests. Studies of nesting success indicate that, for some species, nest success may be higher in the floodplain than in the uplands. Floodplain birds face threats from large-scale loss of forest habitats and their constituent natural levees, backwater sloughs, and oxbow lakes. Conservation efforts should focus on restoring degraded floodplains and maintaining high tree species diversity and wide corridors. To accomplish this, the underlying hydrodynamics which support a diverse floodplain forest habitat may need to be restored. Large, contiguous tracts of floodplain and adjoining upland forests should be maintained where they exist and restored in other locations. Large tracts are needed to reduce the effects of cowbird parasitism and predation and provide some high quality habitat for area-sensitive neotropical migratory birds (NTMBs) in agricultural landscapes where small, scattered forest fragments are the rule. The removal of known cowbird feeding areas within and adjacent to forests will also likely reduce brood parasitism. Primary and secondary cavity nesters make up a significant part of floodplain forest bird communities and managers should give high priority to providing an abundant supply of dead trees and snags. Future research efforts should examine the importance of floodplain forests in maintaining populations of neotropical migrants, especially birds experiencing population declines in adjacent uplands.

INTRODUCTION

The floodplain forests found along the network of major rivers of the Upper Midwestern United States have long provided breeding, migration, and wintering habitat for millions of birds. These forests and their associated wetlands have become increasingly important because of wetland and forest losses throughout the region. Large floodplain forests are distinctive features of the landscape and as dynamic habitats, exposed to frequent disturbances, they provide scarce resources for some species of birds that are less common or absent in adjacent upland habitats. We review the available literature on floodplain forest birds focusing on NTMBs and the breeding bird assemblages of floodplain forests of major

[1] *National Biological Service, Upper Mississippi Science Center, 2630 Fanta Reed Road, La Crosse, WI 54603.*

[2] *Illinois Natural History Survey Division, 607 E. Peabody Drive, University of Illinois, Champaign, IL 61820.*

[3] *National Biological Service, Iowa Cooperative Fish and Wildlife Research Unit, 11 Science II, Iowa State University, Ames, IA 50011.*

rivers in the Upper Midwest. Because so little published information is available about Midwestern floodplain forest birds, we include summaries of data collected in two recent studies from the Cache River in southern Illinois and the Upper Mississippi River in Minnesota, Wisconsin, and Iowa. Full reports of methodology and analyses of these data are being published separately. Our purpose in summarizing this information is to provide a current perspective on Midwestern floodplain forest bird communities and to identify conservation and management needs.

This paper is organized according to the following sections: (1) Dynamics and structure of forests associated with large rivers in the Midwest, (2) The floodplain forest bird community, (3) Potential threats to floodplain forest-nesting birds, and (4) Conservation and management of floodplain forest birds.

DYNAMICS AND STRUCTURE OF FORESTS ASSOCIATED WITH LARGE RIVERS IN THE MIDWEST

The floodplain forests of the Upper Mississippi, much of the lower Missouri, and parts of the Ohio River systems can be grouped into two major forest types (Eyre 1980) (fig. 1). The elm-ash-cottonwood type occupies most of the floodplains of this region (referred to by Eyre as the Central Forest Region) (see Appendix A for scientific names of trees and shrubs). During the last century, as a result of flood control, irrigation, and other water projects, the elm-ash-cottonwood group has expanded westward in distribution along riparian corridors of the Great Plains (Knopf and Scott 1990). This expansion of eastern floodplain forests is responsible for an influx of eastern bird species into eastern Colorado (Knopf 1986).

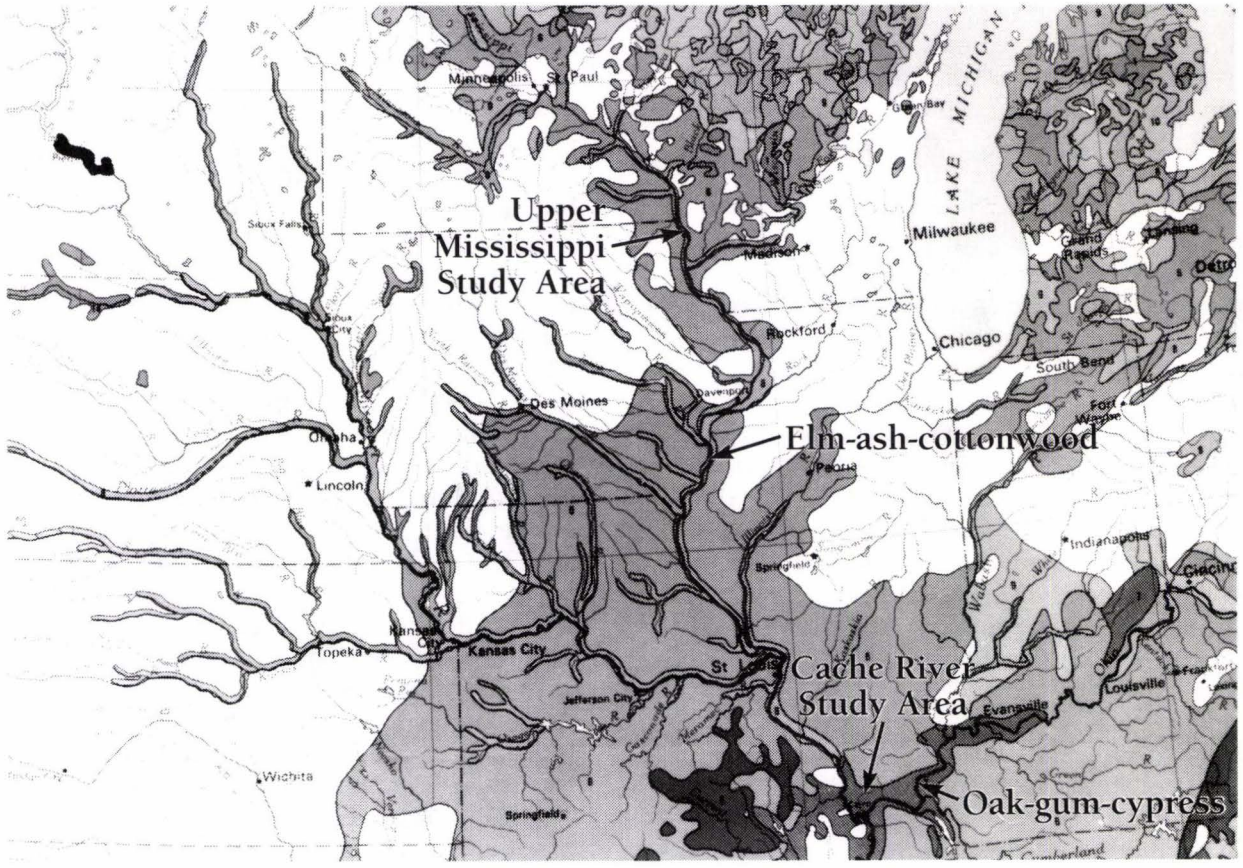

Figure 1.—*Major floodplain forest types (oak-gum-cypress and elm-ash-cottonwood) of the Central Forest Region of the United States. Elm-ash-cottonwood forests are bordered by either oak-hickory forest types (dark shading) or tallgrass prairie (no shading). Excerpted from national Atlas of Major Forest Types with an original scale of 1:7,500,000 (Eyre 1980).*

The oak-gum-cypress forest type of the lower Mississippi River alluvial plain extends into the Central Forest Region for a short distance north of Cairo, Illinois on the Mississippi and up the Ohio River into southern Indiana. The Cache River in southern Illinois enters the Ohio near its junction with the Mississippi and is in a transition zone between the elm-ash-cottonwood and the oak-gum-cypress forest types.

Each of these two general forest types contains 12 or more cover types (species groups) named for the tree species that are dominant in basal area within a stand. The elm-ash-cottonwood type includes: silver maple-American elm, river birch-sycamore, hackberry-American elm-green ash, willow, and cottonwood cover types. The oak-gum-cypress forest type includes: pin oak-sweetgum, swamp chestnut oak-cherrybark oak, sugarberry-American elm-green ash, sycamore-sweetgum-American elm, black willow, and bald cypress-water tupelo. Various species of oak, hickory, pecan, and maple are important associated species in most of these cover types. The composition of forest tree communities changes along latitudinal and climatic gradients, but on a local scale, diversity within a floodplain tree community is influenced by the elevation profile of the river channel from bank to bank. Plant species distributions arise from different tolerances to flooding, with the most tolerant species dominating the lowest elevations. Trees adapted to drier conditions are found on levees and the higher terraces adjacent to the rivers. Plant distributions also depend upon seed survival and dispersal dynamics (Yeager 1949; Hosner 1958, 1960; Galatowitsch and McAdams 1994). Trees with seeds that disperse by wind (cottonwood, silver maple) colonize more readily than trees with larger seeds that depend upon water or animal dispersal (oaks). Seedlings are often more sensitive to the effects of flooding than adult trees.

Large river fluvial dynamics (movements of sediment influenced by river hydrology) create an interspersion of forests, side channels, backwater sloughs, oxbow lakes, and marsh habitats (Malanson 1993) (fig. 2). This mix of habitats may remain stable over periods of 50-100 years within a region, but the location of habitat patches is dynamic because of erosion and land-building associated with fluvial processes (Kupfer and Malanson 1993). Flooding is a natural, periodic disturbance that strongly influences the species composition, size and shape of forested land in the floodplain. Over geologic time, the Mississippi River has served as a corridor for the gradual "migration" of tree species responding to climatic changes, especially the pecan (Bettis *et al.* 1990). The corridor may prove an important north-south link for tree species migration in the event of global warming.

According to the river-continuum concept, riverine forests are connected hydrologically, chemically, and biotically to each other and the larger upland matrix via transport of water and nutrients from uplands to the floodplain and downstream (Naiman *et al.* 1993, Johnson *et al.* 1995). Upstream forests are also connected to downstream forests via water flow. Therefore, riverine processes such as sedimentation (which can ultimately lead to development of forests) are closely tied to land use practices and disturbances to natural land cover in uplands adjacent to and upstream from a particular floodplain forest. In addition, contaminants from agricultural or industrial sources can move through these systems via water flow, especially under flood conditions. Thus, floodplain forests play a critical role in filtering nutrients and sediment and act as a buffer against potential effects on water quality of rivers from disturbances in upland portions of their watersheds (Peterjohn and Correll 1984, Brinson 1990). According to the flood pulse concept, periodic flooding (usually annual) is the most important hydrologic feature of large rivers (Johnson *et al.* 1995). Flood pulses extend the river onto the floodplain and directly affect nutrient availability and aquatic organisms (Junk *et al.* 1989, Power *et al.* 1995). Because some floodplain birds use aquatic insects and fish as sources of food, it is likely that flood pulses are important for them as well.

The complex interplay of these dynamic forces likely plays a major role in maintaining high biodiversity in floodplain corridors (Naiman *et al.* 1993). Riverine trees generally have greater basal area, higher biomass, and more biomass production than adjacent upland forests (Brinson 1990). Plant species richness can vary with the size of the river, with higher richness in the larger rivers than in the tributaries (Nilsson *et al.* 1994). However, within a

Figure 2.—*Aerial view of the Upper Mississippi River floodplain showing multiple channels, backwaters, marshes, forests, and adjacent uplands (left side of highway). The city of LaCrosse, Wisconsin is in the upper background of the photo.*

given floodplain, the number of tree species increases from the most frequently flooded and wetter soils near the main channel to the less frequently flooded soils at higher elevations of the floodplain (Brinson 1990). Species richness of aquatic invertebrates can increase with stream size (Minshall *et al.* 1985). Bird abundance increased with the width of riparian forests in Texas (Dickson *et al.* 1995). Riverine forests in both humid and arid environments, generally support more avian species than do adjacent uplands in the same geographic region (Lugo *et al.* 1990).

THE FLOODPLAIN FOREST BIRD COMMUNITY

Migration and Dispersal

Floodplain forests can have a profound influence on the local and regional bird community by functioning as a dispersal and migration corridor for forest-dependent species, especially in landscapes otherwise devoid of trees, such as the western United States (Knopf and Samson 1994). This may also apply to large areas of the Midwest, where row-crop agriculture dominates the landscape. Emlen *et al.* (1986) described the elm-ash-maple forest of the Upper Mississippi Valley as "...one of the longest north-south stretches of relatively uniform vegetation on the North American Continent."

The importance of the Mississippi River and its tributaries as important migration corridors for birds such as waterfowl is well known (Bellrose 1976). Floodplain forests of this river system may also provide important migratory corridors for neotropical birds that feed and rest in riverine habitats. Data collected by the U.S. Fish and Wildlife Service on the Upper Mississippi River Wildlife and Fish Refuge and the Mark Twain National Wildlife Refuge indicate that NTMBs consistently use the floodplain forests during migration in both fall and spring. In 1993, 150 species of birds were identified in Pools 4-8 during spring migration, 20 percent of all birds counted were NTMBs (Schumacher 1993).

Breeding

In Iowa, floodplain forests host a larger number of breeding bird species than upland forest, cropland, shrubland, or marsh (Best *et al.* 1995). Floodplain forests also rank very high in providing primary breeding habitat for Midwestern bird species experiencing population declines such as the cerulean warbler, prothonotary warbler, Louisiana waterthrush, and northern waterthrush (Thompson *et al.* 1993).

Breeding birds in floodplain forests include species that occupy a wide variety of habitat types, including species that specialize on early successional habitats created by flood disturbances. Floodplains provide habitat for some birds that are uncommon or absent from adjacent upland habitats such as prothonotary warblers and red-shouldered hawks (Murray and Stauffer 1995). In some cases, large trees and snags are more common in floodplains than in adjacent uplands and provide better habitat structure for cavity-nesting species (Renken and Wiggers 1993). The connection of floodplain forests with uplands may play an important role during the breeding season, by buffering isolation effects (Decamps *et al.* 1987) and influencing rates of species turnover (Gutzwiller and Anderson 1987).

Few studies have focused on birds breeding in floodplain forests in the Midwest. Emlen *et al.* (1986) surveyed 12 study sites on the Mississippi River between Cairo, Illinois and St. Paul, Minnesota in June 1979, and correlated latitude, habitat and other environmental factors with population densities for 43 species. The bird community changed in a predictable manner over latitudinal gradients with ecologically similar species replacing each other based on climatic affinities. Concern has been raised about the effects on the bird community from fragmentation and loss of these floodplain forests due to lock and dam construction, agriculture, and urban development (Knutson 1995, Grettenberger 1991). Forest fragmentation has been shown to lead to loss of bird species from upland forests (Robbins *et al.* 1989). Studies of Midwestern riparian bird communities demonstrate high bird diversity present in these habitats. A study of the St. Croix River floodplain, a northern Wisconsin tributary of the Mississippi, showed high species richness, with as many as 155 birds breeding in or near the corridor (Mossman 1991). Small and Hunter (1989) found no differences in abundance or species richness between forest-river edges and the forest interior. Best and Stauffer (1980) detected no differences in nesting

outcome for birds breeding in upland and floodplain woodlands in Iowa but indicated that habitat structure and predation were important factors affecting nest success.

Cache River, Illinois Case Study

This ongoing research by Jeffrey Hoover of the floodplain forests of the Cache River watershed in Southern Illinois includes data provided by Scott Robinson (Illinois Natural History Survey) from a study conducted in the nearby upland forests of the Trail of Tears State Forest in Southern Illinois. The Cache River watershed, located in the southern tip of Illinois, is a breeding, wintering, and migratory habitat for over 200 bird species. The plant communities found in this watershed include barrens, upland and floodplain hardwood forests, and cypress swamps. Some large (> 1,000 ha) blocks of forest habitat remain along sections of the Cache River, which still flows unconstrained through a broad floodplain with natural levees and backwaters. An ambitious forest acquisition and restoration effort encompassing 60,000 contiguous acres of land is now underway in the Cache River watershed.

The composition of the floodplain forest of the Cache River is diverse. Many species of trees are present in the tracts of floodplain forest in this area. The most common trees include oaks (white, Shumard, black, pin, cherrybark, swamp chestnut), maples (black, red, silver, sugar), hickories (shagbark, bitternut, mockernut, pignut, shellbark), American elm, sweetgum, sugarberry, and American hornbeam. Wet areas are composed of baldcypress, water tupelo, and buttonbush. The shrub layer includes spicebush, blackhaw, swamp holly, pawpaw, and greenbrier.

The Trail of Tears State Forest is less complex and diverse than the Cache River floodplains and is dominated by oak-hickory forests (red and white oak) on the hillsides, ridge tops, and coves, with younger mixed hardwoods (e.g., sugar maple, and tuliptree), in wider ravines. The topography of this forest is very uniform and consists of narrow (≤ 50 m wide) ridges and narrow ravines ranging from 800-2,500 m in length. The forest constitutes part of the eastern extent of the Ozark mountains.

Data from the Cache River floodplain forest were collected in 1993 and 1994 and data from the Trail of Tears State Forest were collected from 1990-1992. Presence and abundance of all breeding birds were monitored using standard point-count methods (Hutto *et al.* 1986). Point counts included distance estimates, vocalization type (song versus call) and compass direction for all species seen and/or heard during 6-minute periods. Only birds present within 70 m of the census points were considered for this analysis. A total of 1,495 points were censused in floodplain forest and 1,703 in upland forest. The number of species and relative abundances (mean number of individuals per 10 70-m radius census points) of NTMBs in floodplain and upland forest were compared.

To examine the effect of corridor width on the bird community, birds were censused in 14 forested corridors within the Cache River project area from 20 May to 30 July 1993 and 1994. All of the forested corridors included in this analysis were adjacent to or were bisected by flowing water. The width of forested corridors was determined by averaging the width (in meters) of the corridor at each end of the area that was censused, and the width at two points between each end that were equidistant from each other and the ends. All intensive study plots in which nesting success was quantified were censused 3 times. In these plots, points were located at 150 m intervals. These study plots were designed to sample the small (<100 ha) tracts entirely and to sample representative 100-150 ha portions of the larger tracts.

Nesting success of breeding birds was also monitored in these studies. Nests were located and monitored at 2 to 3 day intervals until they were no longer active. Daily predation rates were calculated for each species using the Mayfield (1975) index. Statistical comparisons of predation levels were made using the program CONTRAST (Hines and Sauer 1989). The frequency of brood parasitism by the brown-headed cowbird was also documented. We compared the daily predation rate and frequency of brood parasitism between floodplain and upland forest for three species of NTMBs (Kentucky warbler, wood thrush, and Acadian flycatcher) that were common in both forest types.

A total of 33 species of NTMBs were identified during point counts in the floodplain and upland forests combined (table 1). The hooded

warbler was the only species present in the uplands not present in the floodplains, although the worm-eating warbler was only present along bluffs adjacent to the floodplain. More species were detected in the floodplain forest (32) than in the upland forest (25). Based on the censuses, seven species occurred only in floodplains (chimney swift, eastern kingbird, eastern phoebe, yellow warbler, prothonotary warbler, American redstart, orchard oriole, and northern oriole); only the prothonotary warbler was common. Some of these species may be present only in the floodplains because they require habitat features not found in the uplands. For example, chimney swifts used large, hollow trees for nesting sites. Eastern phoebes built nests along the steeper banks of the Cache River and along some of the bluffs that overlook the river. The more abundant prothonotary warbler prefers flooded forests and numerous snags with natural cavities for nesting. The American redstart preferred forest-water edges.

Table 1.—*Relative abundances of neotropical migrant songbirds in floodplain and upland forest habitats in southern Illinois.*

Species	Number of individuals per 10, 70-m radius census points	
	Floodplain (1993-1994) (1495 census points)	Upland (1990-1992) (1703 census points)
Yellow-billed cuckoo	4.17	1.62
Chimney swift	2.13	—
Ruby-throated hummingbird	1.65	1.10
Eastern kingbird	0.007	—
Great-crested flycatcher	3.37	2.76
Eastern wood-pewee	7.69	9.23
Eastern phoebe	0.03	—
Acadian flycatcher	19.6	11.0
Blue-gray gnatcatcher	9.60	7.37
Gray catbird	0.19	0.02
Wood thrush	3.18	3.26
Red-eyed vireo	3.21	6.17
Yellow-throated vireo	2.09	1.97
White-eyed vireo	1.14	1.50
Northern parula	3.27	1.26
Yellow-throated warbler	0.61	0.38
Prothonotary warbler	5.12	—
Black and white warbler	0.007	0.02
Cerulean warbler	0.47	0.21
American redstart	0.15	—
Yellow warbler	0.013	—
Worm-eating warbler	0.03	6.88
Hooded warbler	—	0.67
Kentucky warbler	2.91	5.80
Common yellowthroat	1.44	0.01
Yellow-breasted chat	0.23	0.02
Louisiana waterthrush	0.38	0.56
Ovenbird	0.03	2.04
Orchard oriole	0.06	—
Northern oriole	0.07	—
Summer tanager	0.85	2.05
Scarlet tanager	0.57	3.62
Indigo bunting	6.39	2.38
Total neotropical migrants	79.4	71.9

Riverine edge habitats are created by flood waters gradually changing the course of the river, causing it to meander back and forth across the floodplain over time (Kupfer and Malanson 1993). Flooding can also weaken root systems of trees, leading to tree-fall gaps. River edges, particularly where the Cache still flows relatively unconstrained, are favorite habitat for the cerulean warbler. The dense cane stands along the river provide potential habitat for the Swainson's warbler (extirpated since 1992), should it return to Illinois. Forest edges with shrubs provide excellent habitat for the indigo bunting and white-eyed vireo.

Species more abundant in the uplands than in floodplains were the eastern wood-pewee, worm-eating warbler, red-eyed vireo, Kentucky warbler, scarlet tanager, summer tanager and ovenbird (table 1). In the floodplain forest, the Acadian flycatcher, blue-gray gnatcatcher, prothonotary warbler, yellow-billed cuckoo, northern parula, and indigo bunting were most abundant. Water levels throughout the Cache River floodplains vary greatly from place to place and fluctuate over the course of the breeding season (J.P. Hoover, data on file). The ground nesting substrate in floodplains is occasionally and unpredictably under water, limiting habitat availability for ground-nesting and ground-foraging species. This may be the reason ground nesters like the ovenbird, worm-eating and Kentucky warbler are more abundant in upland forests. Worm-eating warblers are also associated with moderately steep forested slopes that border the floodplain in a few places. Kentucky warblers are common in the floodplains, but depend on natural levees and areas of higher ground adjacent to the river, where soil has been deposited by spring floods over the years, for nesting and foraging habitat. Without these levees, Kentucky warblers would likely become rare or disappear. Natural flooding is needed to maintain this special type of habitat. A total of about 80 NTMBs per 10 census points in floodplain and 72 in upland forest habitat were found.

Sufficient numbers of nests were obtained to compare predation rates between upland and floodplain forests for 3 species. Daily predation rates did not differ for the Kentucky warbler or for the wood thrush (table 2). Daily predation rate estimates were 4.35 percent for the Kentucky warbler and 3.95 percent for the wood thrush. These daily rates result in overall predation rates for the nesting period of 62 percent for the Kentucky warbler and 65 percent for the wood thrush. However, the daily predation rate was significantly lower ($P<0.05$, CONTRAST) for Acadian flycatchers nesting in floodplain forests than in upland forests (table 2). Overall rates of predation for Acadian flycatcher nests were 61 percent for floodplain and 76 percent for upland forests.

Table 2.—*Daily predation rates for birds nesting in floodplain and upland forest habitats in southern Illinois.*

Species	Daily predation rate (%) ± SD (exposure days)	
	Floodplain (1993-1995)	Upland (1990-1992)
Kentucky warbler	4.4 ±0.69 (871)	4.3 ±0.82 (607)
Wood thrush	3.7 ±0.33 (3280)	4.2 ±0.56 (1301)
Acadian flycatcher	3.4 ±0.24 (5787)	5.1 ±0.49 (2030)

The frequency of brood parasitism was not different for the Kentucky warbler ($X^2 = 0.84$, df=1, $P>0.1$), wood thrush ($X^2 = 1.58$, df=1, $P>0.1$), and the Acadian flycatcher ($X^2 = 1.09$, df=1, $P>0.1$) (table 3). Floodplain and upland forests of southern Illinois are fragmented by agriculture which provides feeding areas for cowbirds. There are few places (if any) in the forests of Illinois that are far enough away from cowbird feeding areas to prevent brood parasitism. Recent studies in Illinois and Missouri show that cowbirds moved an average of 1.2 km and a maximum of 3.2 km between feeding and breeding areas (Thompson 1994). However, they were capable of moving 10 km or more in a day. A large-scale restoration project such as the Cache River Bioreserve could create blocks of forest extensive enough to reduce the current levels of parasitism. The rates of nest predation and brood parasitism found here are consistent with rates found for these species throughout Illinois (Brawn and Robinson 1996).

Table 3.—*Levels of brood parasitism for birds nesting in floodplain and upland forest habitats in southern Illinois.*

Species	Percentage of nests parasitized (n)	
	Floodplain (1993-1995)	Upland (1990-1992)
Kentucky warbler	36 (87)	43 (75)
Wood thrush	89 (238)	84 (108)
Acadian flycatcher	38 (365)	33 (169)

The correlation for species richness of NTMBs with floodplain corridor width was significant and positive (fig. 3: r^2=0.64, F=21.1, P=0.0006). From a management perspective, data from the Cache River study point to the following relationships: Increasing the width of forested corridors will increase species richness, especially for NTMBs. Additional data indicate that increasing corridor width will increase species richness, with most of the beneficial effect occurring between 200 and 600 m (Robinson and Hoover 1995). The increase in species richness in very wide corridors comes largely from the addition of a transition zone between floodplain and upland forests where increased diversity of vegetation favors a higher diversity of birds. The presence of flowing water in a tract of forest also increases species richness by attracting water birds and species such as the prothonotary warbler that nest at the interface of forest and water.

Results from the first 2 years of the Cache River study suggest that the largest tracts of floodplain forest have the highest diversity and abundance of migrant songbirds, and also contain area-sensitive species such as the cerulean warbler (J.P. Hoover and S.K. Robinson, data on file). Preliminary results also suggest that nesting success of songbirds may be better in wide floodplain forest corridors than in narrow ones (L. Chapa, Illinois Natural History Survey, pers. comm.). This implies that restoration of large blocks of unfragmented floodplain forest along the Cache River will greatly benefit NTMBs in southern Illinois.

Upper Mississippi River Case Study

This study was conducted by Melinda Knutson in the floodplain forests of the Upper Mississippi River from Winona, Minnesota to Prairie du Chien, Wisconsin (Knutson 1995). Census methods were similar to those used in southern Illinois, except point counts were done in 50 m-radius circles for 10 minute periods. For comparisons, data are taken from 700 point counts in the floodplains and 179 in the uplands in 1993 and 1994. Differences in the distribution of counts of individual species (classes of 0, 1, and $\geq$ 2) between uplands and lowlands were tested with chi-square analyses. Additional studies included a pilot study of nesting success conducted in 1992 and an artificial nest study conducted in 1993 and 1994. Methods in the nest success study were similar to those used in the Illinois study. In the artificial nest study, artificial nests containing 2 bobwhite quail eggs were placed about 2.5 m from the ground in trees 25, 50, 100, and 200 m from forest-water edges in the floodplain. The artificial nests were designed to simulate some features of American redstart nests, one of the most common lower canopy nesters in these forests. After 6 days, disturbances (loss or destruction of eggs) were recorded. Large (196-226 ha) and small (70-159 ha) forest tracts were compared, along with distance from edge. Four hundred thirty-one nests, evenly divided between large and small forest tracts were monitored over the 2 years.

Forests in this region are dominated by silver maple, followed by smaller components of green ash, American elm, swamp white oak, cottonwood, and hackberry (fig. 3). The hardwood community (oak, hickory, and walnut), found on slightly drier sites, has been greatly reduced since presettlement times. Surveyor's records indicate that prior to European settlement cottonwood and black willow were more dominant than they are now (Moore 1988). They represent the pioneer community of the floodplain, with seedlings regenerating on open substrates like sand or mud flats. River dynamics have been constrained by the locks and dams since the 1930's, the location of the mature forests has stabilized, and these species have declined (Yin and Nelson 1995). However, mean water levels and the height and severity of flooding have also increased, lead-

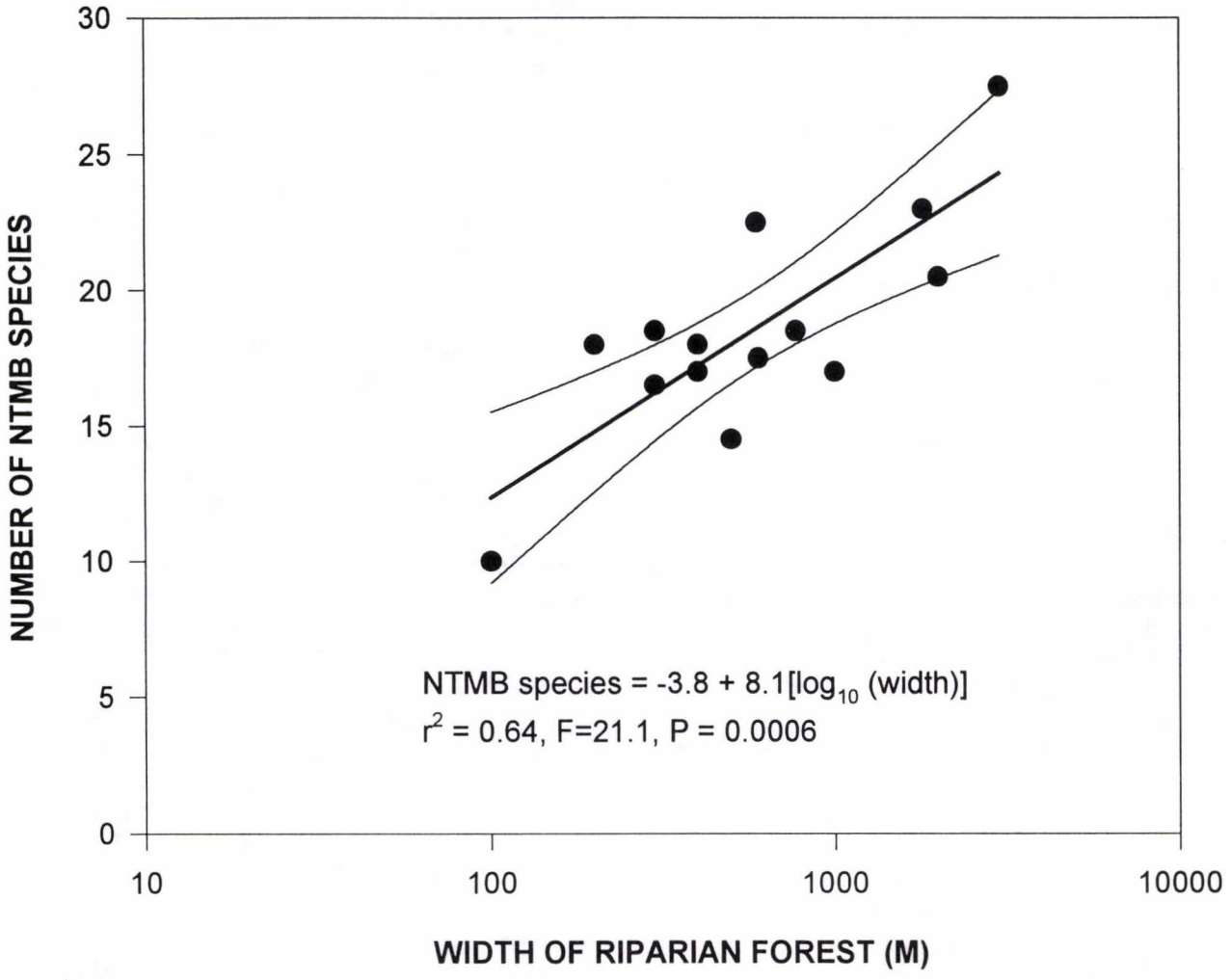

Figure 3.—*Correlation of the number of neotropical migratory bird species (y-axis) with log_{10} corridor width (x-axis) of riparian forest, Cache River, southern Illinois, 1993-1994. The regression line and 95 percent confidence intervals are shown.*

ing to silver maple dominance. Silver maple is more tolerant of flooded conditions than its former co-dominants, American elm and green ash. Hardwoods and other floodplain species, such as hackberry, cannot tolerate long periods of inundation and the new higher water levels (Yeager 1949; Hosner 1958, 1960; Jones *et al.* 1994; Nelson *et al.* 1994; Yin *et al.* 1994). Tree species composition and the natural disturbance cycles have changed as a result of the locks and dams, and these habitat changes may have affected the bird community. Unfortunately, only anecdotal information is available about the bird community prior to the Emlen *et al.* (1986) work in the 1970's.

Knutson detected 84 species of birds during the breeding season, of which 56 are NTMBs. Twenty of the 84 species are cavity-nesters, including seven woodpecker species. Woodpeckers are an important component of the bird community because they create nest holes for secondary cavity nesters, including NTMBs such as the prothonotary warbler, great crested flycatcher, chimney swift, tree swallow, and house wren. Cavity nesters are a conspicuous component of the bird community in the floodplain because of the availability of numerous large standing snags.

A total of 47 species of NTMBs were present in the floodplain, of which 4 (killdeer, cliff swallow, veery, golden-winged warbler, and white-throated sparrow) were not present in uplands. Knutson detected 52 species in upland forests, of which 9 were not found in floodplain forests (turkey vulture, cliff swallow, mourning

warbler, black-and-white warbler, Louisiana waterthrush, Kentucky warbler, rufous-sided towhee, vesper sparrow, and chipping sparrow).

Relative abundances of all birds and total numbers of NTMBs were almost twice as high in the Upper Mississippi River floodplain than in the adjacent uplands (table 4). The most abundant NTMBs in floodplain forests were American redstart, house wren, great crested flycatcher, northern oriole, prothonotary warbler, warbling vireo, and common yellowthroat. The most abundant NTMBs species in upland forests were the ovenbird, red-eyed vireo, rose-breasted grosbeak, scarlet tanager, cerulean warbler, and blue-winged warbler.

Abundance of most floodplain birds did not vary with the amount of forest within 800 m of the census plot. This was true even for American redstarts and blue-gray gnatcatchers, species found to be area-sensitive (show a positive relationship between probability of occurrence, abundance, and size of forest tract) in some upland studies (Stauffer and Best 1980, Temple 1986). The yellow-billed cuckoo, red-bellied woodpecker, yellow-bellied sapsucker, hairy woodpecker, white-breasted nuthatch, and brown creeper were area-sensitive in floodplain forests (Knutson 1995). Knutson's (1995) study included 1993, a year of major flooding for the entire upper Midwest. She examined the effect of the flood on the bird community and found that the relative abundance of some species (blue jay, house wren, and common yellowthroat) was lower the year of the flood and the breeding season following the flood.

In the pilot nest success study, Mayfield nest success (Mayfield 1975) for the American redstart was 25.1 percent (S.E.=8.7 percent, N = 23), prothonotary warbler was 17.7 percent (S.E.=7.5 percent, N = 24), house wren was 34.8 percent (S.E.=10.7 percent, N = 30), and American robin was 15.8 percent (S.E. = 10.4 percent, N = 13) (M. Knutson and E. Klaas, data on file). These estimates are similar to rates of nest survival found by Robinson (1992) in Illinois (abcut 20 percent for NTMBs).

Contrary to many similar studies, in the artificial nest study, large forest tracts had higher predation rates than small forest tracts (52.3 vs. 36.3 percent). There was no significant difference in predation rates among nests placed 25, 50, 100, or 200 m from the forest edge. Calculated artificial nest "survival", derived from observed predation rates in 1993-1994, was similar to natural nest survival for several species for the same study area in 1992. Nest losses did not differ among several habitat variables such as tree species composition, canopy cover, and shrub cover measured at the nest sites.

Species of Management Concern in Floodplain Forests

Based on the two case studies above, NTMBs of high management concern (mean score $\geq$ 3.0 based on Thompson *et al.* 1993) that are closely associated with floodplain forests include the cerulean warbler, prothonotary warbler, Acadian flycatcher, yellow-billed cuckoo, yellow-breasted chat, and great crested flycatcher. NTMBs of medium management concern (mean score of 2.0 -3.0 based on Thompson *et al.* 1993) that are closely associated with floodplain forests include the American redstart, chimney swift, northern oriole, indigo bunting, yellow-throated warbler, gray catbird, least flycatcher, warbling vireo, northern parula, and common yellowthroat. Wide riparian corridors can increase species richness. The fact that floodplain forests are not usually found in large, contiguous stands, but are naturally linear and interspersed by sloughs, oxbow lakes, and marsh habitats does not seem to have serious negative effects on species presence/absence or abundance. Indeed, total bird abundance in floodplain forests was higher than in upland forests. More data on reproductive success from other Midwestern locations is needed to assess how successfully some species are reproducing themselves across the landscape. High predation rates for wood thrushes indicate that this species is at high risk for population declines.

POTENTIAL THREATS TO FLOODPLAIN FOREST-NESTING BIRDS

Floodplain Forest Loss and Fragmentation

Most temperate rivers in the world are constrained from their natural fluvial dynamics by dams or other impoundments (Dynesius and

Table 4.—*Relative abundances of bird species during the breeding season in floodplain and upland forests of the Upper Mississippi River, Winona, MN to Prairie du Chien, WI (1993-1994).*

Species	Number of individuals per 10, 50-m radius census points		
	Floodplain (700 points)	Upland (179 points)	P-value[1]
Turkey vulture	—	0.17	0.008
Red-tailed hawk	0.01	0.06	0.366
Red-shouldered hawk	0.06	0.06	0.599
Killdeer	0.04	—	1.000
Mourning dove	0.30	0.11	0.426
Black-billed cuckoo	0.04	0.06	1.000
Yellow-billed cuckoo	0.87	0.17	0.006
Chimney swift	0.20	0.06	1.000
Ruby-throated hummingbird	0.04	0.11	0.270
Belted kingfisher	0.11	0.11	0.736
Yellow-bellied sapsucker	5.21	1.12	<0.0001
Northern flicker	1.67	1.12	0.339
Eastern wood-pewee	5.69	4.08	0.013
Acadian flycatcher	0.09	0.78	<0.0001
Least flycatcher	0.17	0.06	0.765
Willow flycatcher	0.00	0.11	0.041
Eastern phoebe	0.06	0.22	0.059
Great crested flycatcher	7.90	2.01	<0.0001
Eastern kingbird	0.01	0.06	0.366
Tree swallow	1.56	0.06	<0.0001
Northern rough-winged swallow	0.00	0.34	0.204
Cliff swallow	—	0.11	0.204
Brown creeper	0.53	0.22	0.320
House wren	12.64	2.46	<0.0001
Blue-gray gnatcatcher	4.54	3.91	0.054
Veery	0.20	—	0.258
Wood thrush	0.07	1.45	<0.0001
American robin	7.34	2.96	<0.0001
Gray catbird	3.27	2.18	0.131
Cedar waxwing	0.09	0.22	0.031
Yellow-throated vireo	1.80	1.84	0.158
Warbling vireo	5.93	0.22	<0.0001
Red-eyed vireo	1.99	4.86	<0.0001
Blue-winged warbler	—	0.89	<0.0001
Golden-winged warbler	0.01	—	1.000
Yellow warbler	1.26	0.45	0.018
Cerulean warbler	0.26	1.01	0.0001
American redstart	21.30	4.19	<0.0001
Prothonotary warbler	6.10	0.06	<0.0001
Ovenbird	0.10	5.36	<0.0001
Louisiana waterthrush	—	0.17	0.008
Kentucky warbler	—	0.22	0.002
Mourning warbler	—	0.06	0.204
Common yellowthroat	3.21	0.95	<0.0001
Scarlet tanager	0.07	2.68	<0.0001
Rose-breasted grosbeak	1.29	3.91	<0.0001

(table 4 continued on next page)

(table 4 continued)

Species	Number of individuals per 10, 50-m radius census points		
	Floodplain (700 points)	Upland (179 points)	P-value[1]
Rufous-sided towhee	—	1.23	<0.0001
Indigo bunting	0.09	1.34	<0.0001
Chipping sparrow	—	0.11	0.041
Vesper sparrow	—	0.11	0.041
Song sparrow	8.50	0.61	0.0001
Swamp sparrow	0.03	0.11	0.186
White-throated sparrow	0.03	—	1.000
Brown-headed cowbird	3.96	3.85	0.022
Northern oriole	6.29	2.91	<0.0001
American goldfinch	0.86	2.35	<0.0001
Total neotropical migrants	115.79	63.87	

[1]Probability of obtaining the same or greater difference in the distribution of counts of 0, 1, or ≥2 individuals at each census point between upland and lowland habitats, given the null hypothesis of no difference.

Nilsson 1994). The long-term ecological results of these constraints, such as changes in tree species composition and reduced wildlife habitat, are still under study (Hunt 1988). The results of flood control and other engineering changes to the Lower Mississippi River floodplain have been dramatic. About 80 percent of the original Lower Mississippi River floodplain forests have been lost and the cumulative impacts of the changes in this system are substantial (Gosselink *et al.* 1990). The bird community has also changed in response to these land use changes (Fredrickson 1980, Burdick *et al.* 1989). Species richness and population densities of both wintering and breeding birds have declined with cumulative losses of forest area.

The lock and dam system on the Upper Mississippi River has resulted in a different flow regime than prior to impoundment (Grubaugh and Anderson 1988, 1989; Sparks 1995; Wlosinski *et al.* 1995). Water levels are generally higher over the entire year, flood pulses are higher, and periods of very low flow formerly common in the fall have been eliminated. These human-induced changes in river hydrology have caused major changes in the vegetative cover of the floodplain (Hunter *et al.* 1987; Johnson 1992, 1994; Yin and Nelson 1995). As the navigation dams were constructed during the 1930s, floodplain forests immediately above the dams were cleared prior to flooding. Remaining forests located on higher ground or just below the dams gradually changed in species composition to include species tolerant of flooding and wet conditions. It may take years to detect impacts of hydrologic changes on forests. On the Upper Mississippi River, the forest community has become less diverse and the dominance of silver maple has increased since presettlement times (Moore 1988, Yin and Nelson 1995). Early successional stands of cottonwood and willow have been virtually eliminated and elm, ash, hackberry, oak, and sweetgum are less dominant than in the presettlement forests.

Uplands in the Midwest were formerly covered by either tallgrass prairie or oak-hickory forests maintained by fire. Most of the original area of these natural vegetation types in the central Midwest has been converted to agriculture. In addition, many rivers have been altered by artificial levees constructed to protect large areas of the floodplain from flooding. The forests on lands protected from flooding have been converted to agriculture and urban development. Rivers are dynamic systems, and vegetation and bird communities are likely dynamic as well. Changes in flood frequency and duration of flooding imposed by upstream impoundment (as has occurred on the Upper Missouri, Platte, and Kansas Rivers) can lead to dramatic changes in tree species composition, forest cover, and structure (Knopf and Scott 1990, Knopf and Samson 1993).

In a comparison of floodplain land use before and after European settlement and lock and dam impoundment, Nelson *et al.* (1994) found that 56 percent of the presettlement floodplain at the confluence of the Illinois and Mississippi Rivers was forested, while only 35 percent of the floodplain was forested in 1975. In the Illinois counties of Johnson and Pulaski (the site of the Illinois study), most of the forest is floodplain or swamp forest located within an agricultural matrix. These two counties have experienced significant reductions in forest acreage since 1820 (Iverson 1989). The amount of forest in Johnson County decreased from 92,000 ha to 37,400 ha between 1820 and 1985 (a 59 percent reduction). In Pulaski County, the amount of forest decreased from 53,500 ha to 12,400 ha between 1820 and 1985 (a 77 percent reduction).

Agriculture

Birds living in landscapes dominated by agriculture are subject to a number of threats. Floodplain birds face potential risks from agricultural chemicals used in adjacent upland habitats. These risks are intensified because of the close connection between riparian forests and adjacent croplands. No NTMB species in North America is known to be at risk of extinction because of contaminants. However, individuals and, in some cases, populations may be seriously affected either through direct mortality, decreased reproductive success, or degradation of feeding habitat. Levels of most persistent contaminants in the United States have declined in recent years due to clean-up efforts and control of the release of chemicals (Schmitt and Bunck 1995). Recent work on the Upper Mississippi River (Custer, in press) suggests that organochlorine, mercury, and selenium concentrations in great blue heron eggs were not elevated and have declined since the 1970's (Nosek and Faber 1984). Organophosphorus (OP) and carbamate insecticides, which are less persistent in the environment, are frequently used in agriculture, rangelands, and silvicultural areas (Gard and Hooper 1995) and these compounds continue to cause wildlife mortality (Glaser 1995).

Modern agricultural practices such as drainage and channelization of streams change the hydroperiod of floods by speeding water flow into streams and rivers after rains (Lugo *et al.* 1990). Some of these practices have contributed to major flooding on the Mississippi River in the last 25 years (Belt 1975, Interagency Floodplain Management Review Committee 1994). Agricultural practices that ignore principles of soil conservation lead to accelerated rates of soil erosion. Indirectly, agriculture contributed to the creation of locks and dams on many large rivers by creating demand for quick transport of crops on barges (McLeod 1990, U.S. Army Corps of Engineers 1995). Maintenance of barge traffic requires constant dredging of rivers and dumping of dredge spoil, often in forested areas. The combined effects of changed hydroperiods, high sedimentation rates, and the presence of locks and dams has had profound effects on floodplain habitats in the Upper Mississippi River (Wlosinski *et al.* 1995, Yin and Nelson 1995). These diverse agricultural influences on riparian systems continue to create ecosystem stress and could affect populations of forest-nesting birds in the future (Grubaugh and Anderson 1989, Robinson and Marks 1994).

CONSERVATION AND MANAGEMENT OF FLOODPLAIN FOREST BIRDS

Much of the current management of floodplain forests is, by necessity, adaptive management (Walters and Holling 1990). This involves implementing the best available management option, monitoring the biotic community responses, and changing management options as needed. The U.S. Army Corps of Engineers has been inventorying and mapping thousands of acres of floodplain forest on the Upper Mississippi for over a decade (G. Swenson, U.S. Army Corps of Engineers, Rock Island District, pers. comm.). Their foresters are concerned that the present extensive stands of mature silver maple are even-aged and a healthy distribution of younger trees is missing. As these forests mature, there is evidence that they may be replaced by shrub-scrub habitats, with delayed regeneration of forests. To counteract this predicted outcome, the Corps is harvesting 5 to 10 ha plots from forest stands on a planned 80 to 100 year rotation in the Rock Island District, Illinois. Cutting is done about every 5 years within a stand. A few large trees are left in each clear-cut for use by wildlife and to provide a seed source. Eagle roosts and red-shouldered hawk nests are avoided in planning these cuts. The

Corps is evaluating the effects of their clearcuts on breeding birds by conducting point counts annually.

Biologists from the Fish and Wildlife Service and the National Biological Service have drafted *The Great River Flyway: The Management Strategy for Migratory Birds on the Upper Mississippi River*. The goal of this plan is "...to ensure that habitat quality and availability on the Upper Mississippi River are sufficient to support and enhance optimum populations of migratory birds." The process outlined in this document is presented as a prototype for adaptive management at the landscape scale. The plan provides a logical approach for assessing ecosystem status through the use of species information, habitat definitions, and multi-scale digital land-cover data. The development of databases needed to manage Upper Mississippi River habitats to maintain populations of birds in an ecosystem context are far from complete. But the technology and a plan is available, and the implementation of this plan will help unify floodplain management of birds on the Upper Mississippi River.

Healthy populations of floodplain forest birds obviously require adequate habitat. The research summarized here indicates that floodplain forests provide a different type of habitat than upland forests, as demonstrated by differences in presence/absence and abundance of different bird species. Floodplain forests support higher abundances of birds than upland habitats, in some cases nearly double the abundance. Patterns of area-sensitivity are also different. Changes in flood frequency, duration, or depth imposed by impoundment can lead to changes in tree species composition and alterations in the overstory and understory structure of the forest. For example, large areas of mud flats and sand bars are needed for adequate regeneration of cottonwood and willow. With sustained high water levels created by locks and dams, these habitats remain underwater and little seedling establishment takes place. If these changes are large-scale and unidirectional, they could adversely affect species richness and relative abundance of some floodplain forest-nesting species. For example, species preferring the habitat structure provided by silver maples will likely increase on the Upper Mississippi River and those requiring the structure and/or mast provided by cottonwood, elm, and oak will likely decline. Work is ongoing to define the species associations with these forest types.

Adaptive forest management plans should seek to achieve a healthy age distribution and species diversity of floodplain trees. This might include extensive reforestation on reclaimed agricultural land. If possible, some large standing forests should be selected and preserved as old-growth forests. Regeneration of trees from existing seed sources surrounding clear cuts has not been totally successful. Clearings quickly become overgrown with dense stands of vines and herbaceous vegetation that prevent the establishment of tree seedlings. Control of herbaceous vegetation includes herbicide application, mowing, and disking. Natural forest openings over much of the Upper Mississippi River are dominated by reed canary grass, an aggressive grass which shades out shrub and tree seedlings. Regeneration of floodplain forests will require innovative techniques to control reed canary grass and allow tree seedlings to mature. Restoration of "natural" hydrodynamic patterns, to the greatest possible extent, is the approach most likely to succeed in large scale restoration of floodplain ecosystems. Locks and dams on the Mississippi and Illinois Rivers will remain into the foreseeable future. But, property damage resulting from the flood of 1993 and future major floods may stimulate reclamation of some areas to function as floodplain again. Providing continuous floodplain forest habitat along the Mississippi River and other large river corridors may prove especially important for NTMBs during migration, especially where the corridor crosses intensive agricultural areas devoid of forest cover. Areas like this are found in Illinois and southern Iowa. Regeneration of floodplain forests is being considered in other locations around the world where negative effects of river control measures are acute (Zinke and Gutzweiler 1990). Waterfowl and wading birds are being used to assess the success of restoration of floodplain habitats on the Kissimme River, Florida (Weller 1995). Predicted responses to restoration are an increase in species diversity and relative abundance.

Active management (tree planting, control of invasive plants, manipulation of water levels) of existing and restored forests may be necessary to maximize habitat values as long as

some of the forces which degraded these ecosystems (locks and dams, high sedimentation rates, channelization) remain. Forest management plans that include tree cutting to encourage forest regeneration should also provide standing snags for cavity-nesters. Cutting should be designed to minimize fragmentation of the remaining forests as little as possible because some species, such as cerulean warblers, likely require large areas of mature trees. Emphasis should be placed on restoring and maintaining wide floodplain forests. On small- to medium-sized rivers these forests should be at least 200 m wide; on larger rivers forests should be at least 600 m to several kilometers in width. Larger forests and corridors will increase species richness and potentially reduce brood parasitism by increasing the distance between breeding and feeding areas for cowbirds and also by diluting the parasitism with increased host abundances. The removal of known cowbird feeding areas within or adjacent to floodplain forests will also likely reduce brood parasitism. Management actions such as establishment and maintenance of riparian buffer strips high in the watershed could capture sediment and agricultural chemicals at their source and prevent movement into larger-order rivers, benefitting both aquatic and terrestrial wetland species (Peterjohn and Correll 1984).

Natural levees have high species diversity and high nesting success, and may serve as a refuge for ground-foraging and ground-nesting species during flood events. The restoration of natural flood pulses will help to create or maintain these levees. Restoring natural flood pulses should also reduce flash floods that can destroy nests and decrease sudden low flows that can expose nests to predators such as raccoon (*Procyon lotor*) and mink (*Mustela vison*). Upland forests adjacent to floodplain forests may serve as a refuge for ground-foraging and/or ground-nesting species, especially during a year of frequent or high flooding.

It may be possible to predict the composition and structure of the future floodplain forests based on current hydrodynamic information, observations of change already occurring, and our knowledge of vegetation tolerances to flooding. Forest composition and structure could be important factors in determining the composition of the future bird community. More research is needed to understand how hydrologic changes will drive successional processes in these forests and how forest change will influence the regional bird community.

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Appendix A

Common and scientific names of plant species used in text.

Common name	Scientific name
Alder	*Alder* spp.
Ash, green	*Fraxinus pennsylvanica*
Baldcypress	*Taxodium distichum*
Birch, river	*Betula nigra*
Blackhaw	*Viburnum prunifolium*
Buttonbush	*Cephalanthus occidentalis*
Cane	*Harundinaria gigantea*
Cottonwood	*Populus deltoides*
Elm, American	*Ulmus americana*
Grass, reed canary	*Phalaris arundinaceae*
Greenbrier (catbrier)	*Smilax* spp.
Hackberry	*Celtis occidentalis*
Hickory	*Carya* spp.
Bitternut	*Carya cordiformis*
Mockernut	*Carya tomentosa*
Pignut	*Carya glabra*
Shagbark	*Carya ovata*
Shellbark	*Carya laciniosa*
Hornbeam, American	*Carpinus caroliniana*
Maple	*Acer* spp.
Red	*Acer rubrum*
Silver	*Acer saccharinum*
Sugar	*Acer saccharum*
Oak	*Quercus* spp.
Black	*Quercus velutina*
Cherrybark	*Quercus falcata pagodifolia*
Northern red	*Quercus rubra*
Pin	*Quercus palustris*
Shumard	*Quercus shumardii*
Swamp chestnut	*Quercus michauxii*
Swamp white	*Quercus bicolor*
White	*Quercus alba*
Pawpaw	*Asimina triloba*
Pecan	*Carya illinoensis*
Poison ivy	*Toxicodendron radicans*
Possumhaw (Swamp holly)	*Ilex decidua*
Spicebush	*Lindera benzoin*
Sugarberry	*Celtis laevigata*
Sweetgum	*Liquidambar styraciflua*
Sycamore	*Platanus occidentalis*
Tuliptree	*Liriodendron tulipifera*
Tupelo, river	*Nyssa aquatica*
Walnut, black	*Juglans nigra*
Willow, black	*Salix nigra*

NON-BREEDING SEASON CONSIDERATIONS FOR THE CONSERVATION OF MIGRATORY BIRDS IN THE MIDWEST: POST-BREEDING AND WINTERING PERIODS

J. Faaborg[1], A. D. Anders[1], M. E. Baltz[1], and W. K. Gram[1]

ABSTRACT.—Migratory birds have the potential for population limitation on areas other than the breeding grounds. We review the state of current knowledge on the potential for limitation on wintering grounds and post-breeding habitats, with emphasis on Midwestern species and ecosystems. Although little studied, post-breeding habitat selection within the breeding grounds may be important to offspring survival, as many migrants fledge in June but do not migrate until September. We provide data on movements of wood thrush as an example of problems faced at this time. We also examine the winter ecology of Midwest migrants, including habitats used, adaptations for existence, and problems faced. In all cases, we discuss both the opportunities and constraints faced by Midwest managers concerned with migratory birds.

INTRODUCTION

Classical upland wildlife management has attempted to determine ways in which manipulations raise the density of local populations of target species. The shift to management of the diverse set of neotropical migratory birds (NTMBs), however, requires that we recognize that populations of these species are potentially limited on scales much larger than just local. For this reason, local management practices may or may not result in population increases or in the preservation of target species.

A variety of landscape or larger-scale factors may affect distribution and reproductive success of NTMBs during the breeding season (this volume). These factors make it clear that traits of an area being managed such as size, shape, regional habitat distribution, and distance from population sources put strong constraints on how much success a wildlife manager might expect for a specific area. Successful management of many migratory bird populations may require an understanding of ecological factors extending over an area of hundreds or even thousands of square miles.

[1] *Division of Biological Sciences, 110 Tucker Hall, University of Missouri, Columbia, MO 65211*

Managment of breeding NTMBs over such a large spatial scale is difficult enough, however, these migrant species also have the potential to be limited in other geographic areas. The most obvious of these is the wintering grounds; some species spend as much as 8 months in specific wintering habitat. Loss of such habitat could lead to population declines regardless of management activities or habitat availability on the breeding grounds (Askins *et al.* 1990, Sherry and Holmes 1995).

The time between fledging of young and fall migration has been mostly ignored. For some species in the southern part of the Midwest, a single brood of young may fledge in late May or early June, but parents and young do not migrate until late August or September. In many cases, it appears these birds leave the breeding territory and go elsewhere until migration occurs; if special habitat needs exist, potential limitation also could occur during this time.

Habitat can also potentially limit populations of Midwest NTMBs during migration (Ewert and Hamas 1996). Possible limitation of Midwest-breeding NTMBs during their non-breeding season covers a lot of ground, both actually and figuratively. In some cases, it is possible that Midwestern wildlife managers might be able to adopt practices to aid these migrants

during this period, although it is also obvious that managers can do nothing directly about habitat loss in the neotropical wintering grounds. Our goal in this paper is twofold: First, we survey possible habitat needs of migrants during the non-breeding time spent in the Midwest and offer management suggestions when possible. Second, we survey the current knowledge of winter limitation of NTMBs so that Midwestern managers can better understand how factors thousands of miles from their management areas may affect some of the birds with which they deal during the breeding season.

POST-BREEDING, PRE-MIGRATION HABITAT USE

State of Our Knowledge

Although the period of time between breeding and fall migration may seem trivial and specific habitat needs during this time relatively unimportant, the length of time that an individual spends in pre-migration habitats in parts of the Midwest could amount to several months. The ability to find appropriate habitat during this time could be critical. Many single-brooded NTMBs in the southern Midwest fledge young in early June; most are done breeding by July. Many of these species leave their breeding territories shortly after young are fledged. Most individuals do not reach migration stopover sites until September or October (Able 1972, Rappole and Warner 1976, Buskirk 1980). There is a period of 1 to 3 months when adults and juveniles are not on breeding territories or at migration stopover sites.

Very little is known about habitats used during this post-breeding, pre-migration period. The quality of habitats used during this period is potentially important. Individuals must obtain enough food resources to build fat reserves for migration, and they must do this in areas in which the risk of predation is relatively low (Lima 1986, Moore *et al.* 1993). Thus, the knowledge of and ability to manage for the types of habitats that are suitable during the pre-migration period could have important implications for many NTMB populations.

Little work has been done to assess habitat use or requirements during this time, primarily due to the difficulty in tracking individuals from the breeding territories to pre-migration habitats. Researchers have successfully studied migrant habitat needs at stopover sites by finding areas along known routes that contain hundreds of individuals at one time. Areas that individuals use before migration are not this easily located because birds do not aggregate at resource "islands" but settle at lower densities into habitat types that occur over a wide spatial scale. At this time, these non-breeding birds do not sing, and the habitats in which they occur are heavily vegetated, making detection very difficult. Only by intensively censusing the myriad habitat types available or by following individuals from breeding territories can we gain knowledge of pre-migration habitat requirements.

The only research on post-breeding ecology of songbirds is on survivorship and dispersal of winter-flocking residents and short-distance migrants (Dolnik and Blyumental 1967, Holleback 1974, Dhondt 1979, Ketterson and Nolan 1982, Sullivan 1989). Studies of habitat management on the breeding grounds have focused entirely on impacts on breeding habitats (Crawford *et al.* 1981, Freedman *et al.* 1981, Thompson *et al.* 1993a). The only information published on the pre-migration habitat use of a neotropical migrant species is an anecdotal description of four female hooded warblers (*Wilsonia citrina*) using dense shrub habitat after breeding (Morton 1990). Researchers have observed Kentucky warblers (*Oporornis formosus*), ovenbirds (*Seiurus aurocapillus*), and worm-eating warblers (*Helmitheros vermivorus*) in group-selection cuts and young clearcuts at the end of the breeding season (S. Robinson, pers. comm.; A. Anders, pers. obs.), but such observations have not been quantified.

Case Studies

We know of three studies that are currently being conducted to assess post-breeding, pre-migration habitat use of a forest interior NTMB, the wood thrush (*Hylocichla mustelina*). One study is being conducted in lowland beech-oak-maple forest in the Piedmont region of Virginia (J. Vega and W. McShea, pers. comm.), the second in mixed pine/hardwoods in Georgia (J. Lang and M. Conroy, pers. comm.), and the third by one of the authors (A. Anders) in oak-hickory forest in the Ozark Mountains of southern Missouri.

The wood thrush is of particular concern recently because of long-term population declines throughout its range. Of the 110 species of neotropical migrant birds that breed in the Midwest, the wood thrush has been ranked eighth in terms of management concern (Thompson *et al.* 1993b). The wood thrush is particularly conducive to studies of pre-migration habitat use because of its relatively large size. With an average adult body mass of 47.4 g (Dunning 1993), fledglings and adults can be radio-tracked.

Preliminary research on juvenile wood thrush at Peck Ranch Conservation Area in southern Missouri indicates that pre-migration habitat differs markedly from breeding habitat. We followed 12 of 49 juveniles from their natal territories to pre-migration habitats. Juveniles travelled 0.9 to 4.1 km from their natal territories. Habitats used included sapling-sized stands of oaks or pine, forest/field edges, mature wooded riparian habitat, and high-graded forest on private land. In all cases, understory and ground cover were dense, and the birds spent most of their time on or near the ground. We observed study individuals using these habitats for periods of up to 29 days. There were no mortalities of study individuals in any of these pre-migration habitats.

A typical example of pre-migration movement and habitat use by our study birds follows. One juvenile bird remained on the natal territory for 20 days after fledging. This territory was in 111-year-old oak-hickory forest, and the total area used by the family group after fledging was approximately 15 ha. After leaving the natal territory, the juvenile bird moved southwest, passing through mature oak-hickory forest, across a strip of 65-year-old wooded riparian habitat, and back into mature oak-hickory forest. The individual finally settled into a 13-year-old pine regeneration stand approximately 3.5 km southwest of its natal territory (fig. 1). The total area used in this habitat was approximately 3 ha, and the bird remained in this area at least until the transmitter battery died, at which time the bird had been out of the nest for 33 days.

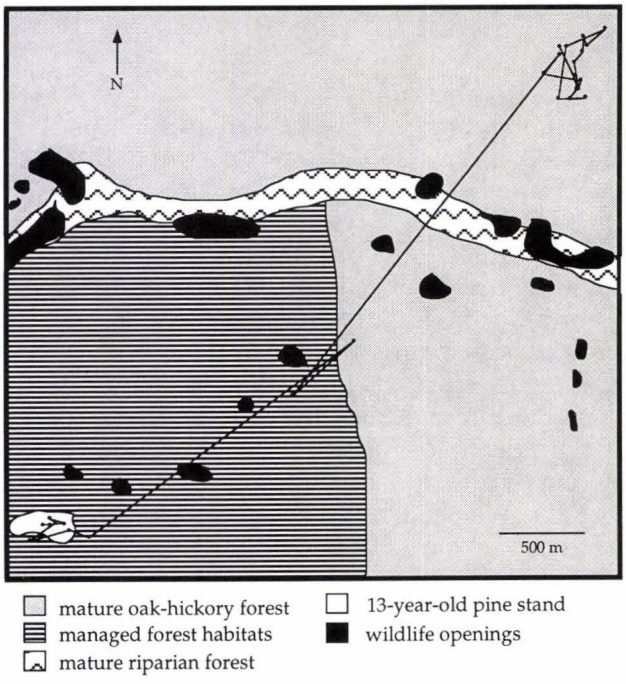

Figure 1.—*Movements of a juvenile wood thrush from breeding habitat to pre-migration habitat in the Ozarks of southern Missouri. Points indicate daily locations from 29 June to 3 August 1994.*

Conservation Considerations for Pre-migration Habitat

Pre-migration habitat of NTMBs may differ from nesting habitat. A landscape-level approach should be used to provide the required interspersion of nesting and pre-migration habitat. We are just beginning, however, to understand how pre-migration habitat may differ from nesting habitat. Further research on movement patterns, habitat use, and survival of individuals during this period is needed to identify critical habitats.

Early successional habitats may play an important role in the pre-migration ecology of NTMBs. The potential exists, however, for increased nest predation and brood parasitism in areas in which forest management is occurring (Thompson *et al.* 1996, Howe *et al.* 1996). However, if nest predation and brood parasitism do not threaten the breeding success of a population, a mosaic of managed forest habitats within large tracts of mature forest, as is found in our study area, may provide important pre-migration habitats for NTMBs.

NEOTROPICAL MIGRANT BIRDS DURING THE WINTER

Species that breed in the Midwest and central Canada tend to have winter ranges that are confined to Mexico, Central America, and the West Indies (fig. 2). The highest densities are in Mexico and the Greater Antilles, with few NTMBs wintering in the expanses of South America (Greenberg 1992). NTMBs annually spend up to 8 months on the wintering grounds, during which time survival and self-maintenance are the primary goals. Survival depends on the availability of suitable wintering habitat that provides predictable food resources and a low risk of predation. When suitable wintering habitat is not available, birds may exploit other, suboptimal habitat with reduced resources, increased predation, and lower survival rates (Winker *et al.* 1990). For NTMBs that occupy a variety of habitats in winter, determining which are the best habitats for a species will require detailed research involving long-term measures of survival and fitness (Conway *et al.* 1995). Because this research is not easy and has not often been attempted, it is not surprising that the importance of winter limitation to declining NTMB populations is controversial (Rappole and McDonald 1994).

What Factors Limit Birds in Their Wintering Habitats?

The most obvious constraint is availability of adequate food resources, which are often linked to particular habitat types. Just as species are adapted to particular habitats during the breeding season, many migrants have specific resource requirements and foraging behavior during the nonbreeding season. While some NTMBs winter predominantly in undisturbed forests, other use and may prefer non-forest and disturbed areas, including agricultural areas (Finch 1991, Hagan and Johnson 1992). Several migrant species are known to switch from forested habitat in the summer to scrub habitat in the winter, or vice versa (such as the least flycatcher (*Empidonax minimus*) and chestnut-sided warbler (*Dendroica pennsylvanica*)). Some species, such as the Nashville warbler (*Vermivora ruficapilla*), are highly flexible in their habitat use during the winter (habitat generalist) whereas other species are limited to a specific habitat (habitat specialist), such as the Kentucky warbler or wood thrush. Other species only winter in a small geographic area or restricted elevational range, such as the Cerulean warbler (*Dendroica cerulea*). There is concern for several Midwestern NTMBs due to wintering ground problems (table 1).

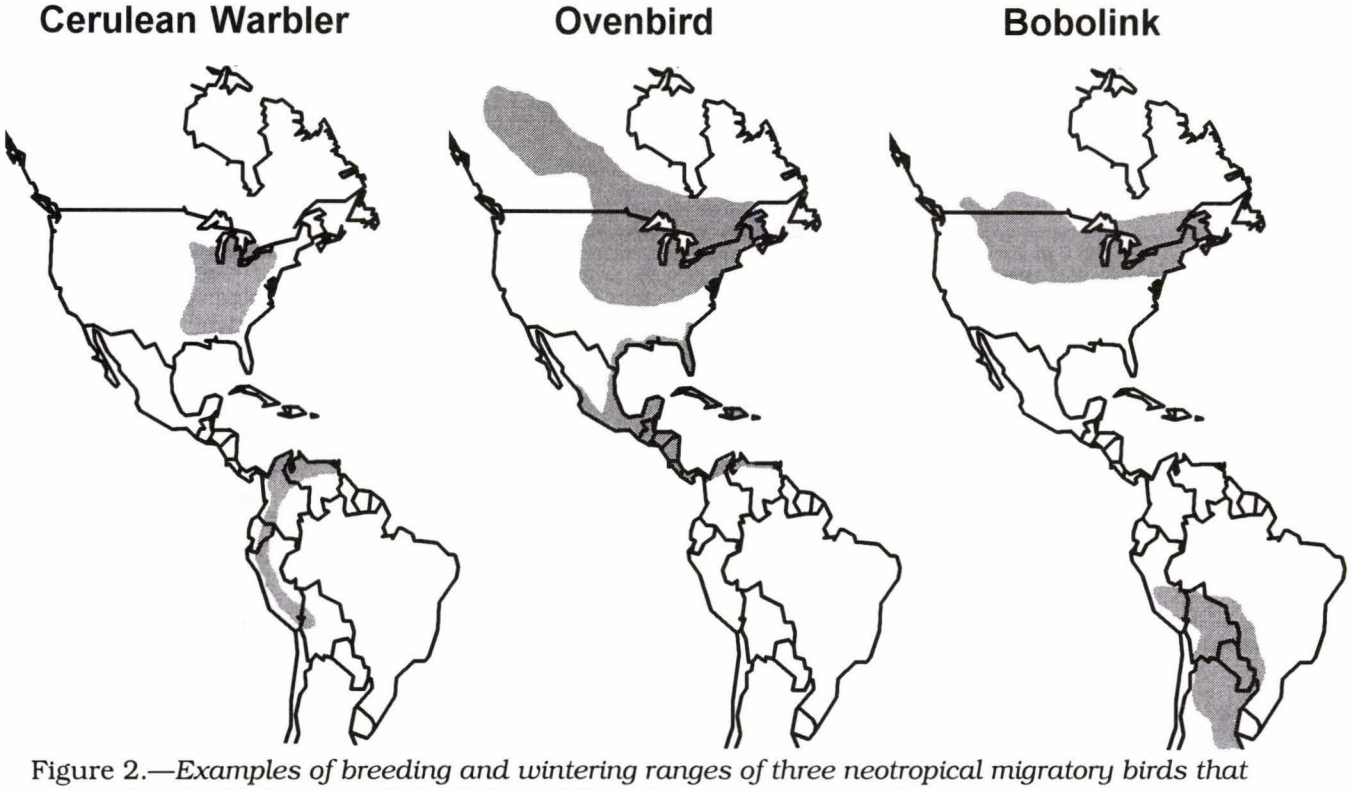

Figure 2.—*Examples of breeding and wintering ranges of three neotropical migratory birds that breed in the Midwestern U.S. (Adapted from Rappole* et al. *1995.)*

Table 1.—*Several midwest breeding neotropical migratory bird species that may be threatened by events on the wintering grounds. Key references are Askins* et al. *1990, Finch 1991, and Greenberg 1992.*

Species	Winter Range	Reason for Concern
Swainson's hawk (*Buteo swainsoni*)	mainly South America	Winters primarily in pampas of southern South America. Sensitive to conversion of grasslands to agricultural use.
Upland sandpiper (*Bartramia longicauda*)	South America	Winters primarily in grasslands of central South America. Sensitive to conversion of habitat to agricultural use.
Olive-sided flycatcher (*Contopus borealis*)	South America	Winters chiefly in mountains of western and northern South America.
Veery (*Catharus fuscescens*)	South America	May winter in a relatively small, rapidly developing area in Brazil.
Wood thrush (*Hylocichla mustelina*)	Mexico and Central America	Moist tropical forest habitat specialist.
Cerulean warbler (*Dendroica cerulea*)	South America	Winters in forest along a narrow altitudinal range in the Andes.
Kentucky warbler (*Oporornis formosus*)	Mexico, Central and South America	Moist tropical forest habitat specialist.
Kirtland's warbler (*Dendroica kirtlandii*)	Bahamas	Only known to winter on a few islands in the Bahamas.
Bachman's warbler (*Vermivora bachmanii*)	Cuba	Believed to have gone extinct due to loss of winter habitat in Cuba.
Prothonotary warbler (*Protonotaria citrea*)	Mexico, Central and South America	Prefers wetlands and mangroves in winter. Uncommon in other habitats.
Northern waterthrush (*Seiurus noveboracensis*)	Mexico, Central and South America, West Indies	Occurs in high densities in mangrove swamps in winter. Uncommon in other habitats.
Dickcissel (*Spiza americana*)	mainly South America	Occurs in large flocks on croplands in winter. Susceptable to changes in agricultural practices on grasslands of South America.

There is an increasing probability that some neotropical migrants will be constrained by the availability of winter habitat as native tropical habitats are converted to agriculture and other human-dominated habitats. Midwestern examples of this are the probable extinction of the Bachman's warbler (*Vermivora bachmanii*) due to loss of its Cuban wintering grounds (Terborgh 1989) and, more recently, concern for the cerulean warbler due to loss of its limited Andean wintering range (Robbins *et al.* 1992).

The social structure of the winter community is also an important aspect of winter habitat use by NTMBs. Many species are very territorial or site faithful in winter, with individuals returning to the same small wintering place year after year (Faaborg and Winters 1979). Several species segregate habitat by sex; in some, dominant males may defend a territory in one habitat with less dominant birds (females and young males) defending less optimal territories or "floating" within the population (Lynch *et al.* 1985, Sherry and Holmes 1995, Conway *et al.* 1995). In some species, habitat selection may differ between males and females (Morton 1990). Other species, such as black-throated green warblers (*Dendroica virens*) and solitary vireos (*Vireo solitarius*), participate in mixed-species foraging flocks composed of both migrant and resident species on the wintering grounds (Gram 1996). Competition with resident species or with similar migrant species may also play a role in determining optimal habitat or winter range.

Although habitat fragmentation has had profound effects upon the nesting success of NTMBs in the Midwest, it is less clear that habitat fragmentation is detrimental to migrants in the winter. In an extensive survey in Central America and the West Indies, Robbins *et al.* (1987) found densities and return rates of migrants to be independent of habitat size, although they found severe fragmentation effects upon the diversity of tropical resident species. Studies of shade coffee forests in the Dominican Republic (Hagan and Johnston 1992; J. Wunderle, pers. comm.) also suggest no effect of forest size on winter residents. Only the work of Askins *et al.* (1992) in the Virgin Islands has shown a negative response of wintering migrants to forest fragmentation, in this case in remnant habitats on the highly developed island of St. Thomas. Given that the negative effects of fragmentation during the breeding season seem to occur with regard to nesting success rather than survival of breeders, perhaps it is not surprising that wintering migrants concerned only with survival may not be area sensitive, although more research is necessary in this area. Additionally, the strong reduction in resident diversity on habitat fragments may actually make life easier for competing species of migrants.

Varied patterns of habitat use reflect the wide diversity of neotropical migrant species and emphasize the conclusion that there are many successful alternative wintering strategies. The ultimate threat occurs when habitats are dramatically altered or disappear completely, leaving species without an adequate supply of resources for winter survival. Although limited data exist on the specific wintering habitat preferences of many species, it is clear that some species could be experiencing a significant reduction in optimal wintering habitats. As we collect more site-specific data on habitat requirements, geographical distribution, and behavior patterns, we hope to link declining population trends that are winter related with conservation solutions.

Land Use Trends in the Neotropics

An estimated 5-10 billion NTMBs of over 150 species squeeze into Mexico, Central America, South America, and the West Indies during the wintering season (Greenberg 1992). Migrant densities are high during the wintering season and it has been estimated that losing a hectare of tropical forest is equivalent to losing 5-8 hectares of temperate deciduous forest (Terborgh 1980). Obviously, the effect of habitat loss on migrant populations in the future could be extreme. For instance, Myers (1991) estimated that the current rates of deforestation in Mexico, Central America and northern South America are 7,000, 3,300, and 63,000 km^2/year, respectively. Based on Terborgh's estimate, this would have the same effect on migrant populations as losing temperate forests the size of Missouri, Illinois, and Iowa annually. Species that have restricted winter ranges or that are uncommon in disturbed habitats are most likely to be affected by this habitat loss. As neotropical areas continue to be destroyed by human alterations for agriculture, logging, and urbanization, suitable wintering habitat for migrant and resident bird species is disappearing.

Of all the habitats in the Neotropics that are experiencing reductions, tropical forest is perhaps the most important to migrants. Over 40 percent of all Midwest migrants commonly occur in or are restricted to this habitat type during the winter season (Rappole *et al.* 1983). Yet, rates of forest conversion range from 1 to 4 percent annually (Gradwohl and Greenberg 1988) and experts predict that by early in the next century the only remaining undisturbed forests in the Neotropics may be in parks or reserves (Raven 1989, Myers 1989). Mangroves and grasslands are also threatened habitats in the Neotropics (Leonard 1987). Although only a small percentage of Midwestern migrants occur commonly in these habitats, several are found almost exclusively here during the winter season (including the northern waterthrush (*Seiurus noveboracensis*) and prothonotary warbler (*Protonotaria citrea*) in mangroves and upland sandpiper (*Bartramia longicauda*) and Swainson's hawk (*Buteo swainsoni*) in grasslands).

Although the future for some migrants based on the rates of habitat conversion may be gloomy (Morton and Greenberg 1989), many species occur in a wide variety of habitats during the winter season. Given the ability of many migrants to use several habitat types, it has been suggested that in the face of major habitat conversions in the Neotropics, relatively few neotropical migrant species are likely to become extinct due to loss of suitable winter habitat (Terborgh 1980). However, many species that are common today may become uncommon or rare in the future, independent of our efforts on Midwestern breeding areas.

Management of Migrant Birds on the Wintering Grounds

What is the best management strategy for NTMBs in their wintering habitat? Just as there are many alternative wintering strategies, there will not be a single best management strategy for NTMBs (Monkkonen *et al.* 1992). It is important to note that Latin American and West Indian land managers must focus their concerns on resident species (especially endemics) and intratropical migrants in addition to the Nearctic-Neotropical migrants that are the focus of this symposium. A tropical manager may have to balance the needs of hundreds of species of birds with a wide range of movement patterns, most of which we do not understand (Levey 1994).

Despite the differences between tropical and temperate systems, exchanging ideas and information on the management of birds in general will undoubtedly contribute to further understanding about the limitations affecting different species throughout their lifetimes. Learning about conservation problems and solutions in other regions of the world may spark new ideas for solutions at home. A detailed understanding of population limitation for a species may prevent managers anywhere in the New World from wasting effort on a species whose actual demise is occurring elsewhere, although it will undoubtedly be a long time before scientists are comfortable enough with our knowledge of any species to support such decisions.

One component of a conservation program for NTMBs on the wintering grounds involves protected lands such as national parks and nature reserves. These will be critical for tropical species that require large areas of undisturbed habitat and for those NTMBs that require mature habitats. The development of biosphere reserves in the tropics should also greatly benefit migrants. These reserves consist of a central protected zone, a surrounding buffer zone with minimal renewable resource exploitation, and an outer zone where additional resource exploitation is permitted. These areas serve as a nice compromise between preservation and development. Temperate managers can encourage preservation of such nature reserves in tropical America through participation in international nongovernmental agencies and with the appropriate lobbying of governmental agencies.

In the face of major habitat conversion, we also need to explore "better-than-nothing" conservation efforts in the Neotropics (Greenberg 1992). Modifying agriculture and logging practices can help make the best of a bad situation for many migrant and resident birds while accomodating the development needs of tropical nations. For example, many studies have shown that leaving small wooded areas, hedgerows, specific fruit trees or roost trees amidst croplands can provide valuable wintering habitat for some migrants and residents (Hagan and Johnston 1992). Robbins *et al.* (1992) reported that arboreal agricultural habitats such as pine, cacao, citrus, and shade coffee plantations support large numbers of NTMBs in some areas,

whereas croplands and overgrazed fields support relatively few bird species unless they contain some wooded areas. Another exciting development is a renewed interest in tropical forest management (Gradwohl and Greenberg 1988, Hartshorn 1992). This approach has many components common to forest management in temperate zones, including selective cutting, reforestation, and research in potentially useful and valuable tree species, all aimed at sustainable use of the forests (Landis 1990, Hartshorn 1990, Tosi 1982).

Although Midwestern managers do not control the wintering habitats of NTMBs, they can encourage and facilitate the exchange of knowledge, data, experience and training with managers who do work directly in tropical habitats. Cooperative research projects, shared training workshops, and the adoption of sister forests are all examples of programs that have furthered the relationship among regions that manage the same migrant bird species. The combined effort of many people with a variety of perspectives is likely to yield timely and creative solutions to the conservation issues facing neotropical migrant and resident bird species.

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ECOLOGY OF MIGRATORY LANDBIRDS DURING MIGRATION IN THE MIDWEST

David N. Ewert[1] and Michael J. Hamas[2]

ABSTRACT.—Insectivorous landbirds migrating through the Midwest during spring and fall encounter a wide array of habitats. Habitat selection by migrants, which appears to be largely driven by food availability and shelter, varies seasonally and geographically. Spring migrants may be relatively food-limited compared to fall migrants, especially in northern portions of the Midwest; emerging aquatic insects provide an important food resource before foliage appears. Further, some migrants have narrow habitat requirements during migration, such as the northern waterthrush (*Seiurus noveboracensis*), or prefer to forage on particular tree species, which may further limit availability of high-quality stopover sites. Because the midwestern landscape has been drastically altered since European settlement, maintaining the quality of remaining stopover sites is critical if birds are to reach their breeding or wintering grounds in good physiological condition. Management and protection of stopover sites is needed for long-term conservation of migratory birds. We offer recommendations directed at protecting and managing sites for migrants. Much remains to be learned; further evaluation of factors governing migratory bird use of stopover habitats is needed.

Key words: food availability, habitat management, landscape, midwest, migratory birds, phenology, stopover sites.

A flood of landbirds moves through the Midwest during spring and autumn migrations when they temporarily inhabit stopover sites as diverse as grasslands, shrubby areas, old-growth forests, urban parks and even ships on the Great Lakes (Perkins 1964). Yet, the ecology of birds during migration is poorly understood. Just as adequate breeding grounds and overwintering areas are requisites for maintenance of migratory bird populations, so too are stopover sites, which provide critical resources necessary for successful seasonal passage. We review habitat requirements of migrant landbirds and provide recommendations for the protection and management of stopover sites in the Midwest to complement the design of hemisphere-wide conservation initiatives (see Moore and Simons 1992, Moore *et al.* 1993). We focus on stopover sites in eastern North America, especially the Midwest, that are used by insectivorous migrant landbirds in forested or coastal sites. Little is known, however, of stopover requirements for grassland (see Askins 1993), frugivorous, granivorous or omnivorous migrants.

[1] *Present address: The Nature Conservancy, 2840 E. Grand River Ave. Suite 5, East Lansing, MI 48823.*

[2] *Present address: Central Michigan University, Department of Biology, Mt. Pleasant, MI 48859.*

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OVERVIEW OF IMPORTANCE OF STOPOVER SITES TO MIGRANTS

High energy requirements associated with migration, approximately 1 percent weight loss per hour of flight (Alerstam 1990), coupled with migrants' exposure to unfamiliar or changing environments en route, require migrants to assess food availability, predators, and shelter at each stopover site (Moore and Simons 1992). Individuals may use the same stopover sites each year although site fidelity appears to be low (Winker *et al.* 1991). Further, different routes may be taken in spring and fall by different species, populations, ages or sexes (Phillips 1951, Crawford and Stevenson 1984, Ramos 1988, Winker *et al.* 1992a). Because habitat selection by each species of migrant may depend upon or change with food availability (Laursen 1978, Bairlein 1983, Hutto 1985, Martin and Karr 1986, Blake and Hoppes 1986, Moore *et al.* 1990, Lindstrom 1989, Sealy 1988, Sealy 1989), intraspecific competition (Rappole and Warner 1976), interspecific competition (Laursen 1978), or habitat availability in different landscapes (see Morse 1980, Moore *et al.* 1990), assessment of the relative value of different habitats to migrants is challenging. However, the ability of a migrating bird to stop in high quality habitats (i.e., habitats with abundant food, low risk of predation and minimum environmental stress) determines whether it reaches the breeding and wintering grounds in good physiological condition. Recognizing these important components of fitness, we can apply our knowledge of habitat use at stopover sites to develop better conservation programs (Moore and Simons 1992). Moore and Simons (1992) succinctly expressed this concern: "Unless habitat requirements during migration are met, conservation measures which focus on temperate breeding grounds and/or Neotropical wintering areas will be compromised."

Migrants in the Midwest

Presettlement landscapes in the Midwest were dominated by a mosaic of forests, wetlands, and grasslands. Habitat loss and fragmentation have drastically altered the landscape. Migrant passage through the region, already made hazardous by storms over the Great Lakes (see Peet 1908, Janssen 1976), must be more difficult than it was 200 years ago when large-scale alteration of the landscape began. Exceptions may occur in parts of Iowa and Minnesota, for example, where forest development in grassland landscapes may benefit some migrants (Faaborg, pers. comm.). Within these landscapes many factors determine which habitats are selected by migrants. Macroscale effects, such as weather, ecological barriers, and habitat availability, may determine when and where a migrant first settles; microscale effects, such as food availability, inter-and intraspecific competition, shelter, and predation, will determine how a migrant fares at a site or within a habitat.

LANDSCAPE ASPECTS OF HABITAT SELECTION

Migrants often accumulate near ecological barriers (Moore and Simons 1992). In the Midwest, particularly large concentrations of migrant landbirds occur along the Great Lakes' shorelines (Peet 1908, Beebe 1933, Tyrrell 1934, Bennett 1952, Weir 1972, Hussell *et al.* 1992, Brock 1992). Some areas have concentrations of many species of birds during both spring and fall migration while other sites are frequented by birds consistently during one season or only periodically. Nevertheless, probably every habitat-type bordering the Great Lakes is used by migrants to some extent during seasonal passage. Habitats adjacent to inland lakes, major rivers, urban areas and large expanses of agricultural land could also have relatively high concentrations of migrants, but these relationships are little known.

Phenological development of vegetation, which is weather mediated, may provide macroscale cues to migrants selecting habitat. In spring, migrants are likely to encounter increasingly unpredictable and sometimes harsh conditions (Blake and Hoppes 1986), as well as delayed phenological development in vegetation, as they proceed north (compare Graber and Graber 1983 with Keast 1980; see Slagsvold 1976). In contrast, fall migrants are likely to have completed their migration prior to first frosts (Bennett 1952) and before leaves have dropped from trees. This interaction between weather and development of vegetation, which is modified by large landscape features like the Great Lakes, results in inter-seasonal differences in habitat choice by migrants (Ewert and Hamas, unpubl. data).

Relationships between patches of habitat in a landscape mosaic and migrant distribution in the Midwest are best described from South Dakota. In this agricultural landscape, Martin (1980) found that the total number of species and abundance of spring migrants was highly correlated with area of small, isolated, homogeneous shelterbelts, but the density of birds feeding primarily in shelterbelts was independent of area. He also concluded that the degree of isolation of shelterbelts did not influence migrants' use of these areas. In unfragmented habitat on barrier islands along the Gulf Coast of Mississippi, Moore *et al.* (1990) noted that migrants moving among heterogeneous patches of habitat preferred shrub-scrub, suggesting that migrants assess and select different habitats where they are contiguous. It is not clear, however, if large patches are consistently more heterogeneous than small patches (Askins *et al.* 1990). Little is known about how or why migrants select a habitat patch or what dispersal occurs among habitat patches.

MICROSITE SELECTION

Food availability may be a primary factor in microhabitat selection within a habitat (Moore *et al.* 1993) and will be further constrained by flexibility in foraging behavior (Robinson and Holmes 1982, Greenberg 1983, Martin and Karr 1990, Parrish 1995), species-specific morphology (Moermond 1990), vegetation structure (Robinson and Holmes 1982), abundance of potential prey (Robinson and Holmes 1982), interspecific competition (Laursen 1978), and intraspecific competition (Rappole and Warner 1976). Because food availability varies among and within habitats, seasonally as well as annually (Laursen 1978, Busby and Sealy 1979, Bairlein 1983, Hutto 1985, Guinan and Sealy 1986, Moore and Yong 1991), migrants depend on a wide array of habitats over time. Yet, given this kaleidoscope of variability, there are some predictable foraging sites or microhabitat features selected by each migrant species for at least portions of migration (Parnell 1969, Power 1971, Rappole and Warner 1976, Bairlein 1983, Winker *et al.* 1992b). Some migrants appear to choose habitats, if available, with vegetation structure similar to that selected during the breeding season (e.g., MacArthur 1958, Parnell 1969). Examples include the northern waterthrush (*Seiurus noveboracensis*), which consistently selects wetlands at stopover sites (Parnell 1969, Rappole and Warner 1976, Winker *et al.* 1992b) and during the breeding season (Bent 1953), and the yellow-rumped warbler (*Dendroica coronata*), which uses a wide range of vegetation in many different habitats during migration (Parnell 1969) and the breeding season (Morse 1989).

Forests comprised of certain tree species and structure may be especially attractive to migrants. Gaps in a deciduous forest in central Illinois consistently attracted migrants during spring and autumn, including foliage-gleaning insectivores, presumably because many insects are available in areas with dense foliage (Martin and Karr 1986, Blake and Hoppes 1986). Moreover, some tree species support relatively large populations of foliage insects and are therefore attractive to migrants after these species have leafed out. Among these tree species are oaks (*Quercus* spp.) (Graber and Graber 1983), yellow birch (*Betula allegheniensis*) (Holmes and Robinson 1981), hackberry (*Celtus laevigata*) (Moore and Yong 1991) and perhaps blue beech (*Carpinus caroliniana*) (Parnell 1969). Other tree species may be avoided as foraging sites by some species and not others (see Robinson and Holmes 1984). These include sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*) (Holmes and Robinson 1981) and sassafras (*Sassafras albidum*) (Graber and Graber 1983).

A stopover site may be more or less attractive to migrants within and between seasons. For insectivores, prey may be scarce when birds first arrive in spring (Nolan 1978, Busby and Sealy 1979, Martin 1980, Guinan and Sealy 1987, Gray 1993, Martin and Karr 1990), forcing migrants to search for sites wherever prey is available (see Keast 1980). Prey-rich sites at this time include areas with high concentrations of geometrid caterpillars on hackberry in Louisiana (Moore and Yong 1991) and oak in southern Illinois (Graber and Graber 1983). Farther north, in the Midwest and in Europe, where migrants encounter less phenologically advanced vegetation, midges (Diptera: Chironomidae) and other insects emerging from water may provide critical food resources, especially early in spring (Laursen 1978; Busby and Sealy 1979; Hamas 1982; Ewert and Hamas, pers. obs.). Thus many migrants and newly arriving summer residents

may concentrate near water (Petit 1991, see Weisbrod *et al.* 1993) while other habitats are avoided. The result is patchy use of the landscape by migrants.

Use of foliar insects increases (Petit *et al.* 1990) as leaves emerge. This may vary by 15 to 20 days from year-to-year at a single site (Nolan 1978), but is approximately early April in Tennessee (Petit *et al.* 1990), mid- to late-April in southern Indiana (Nolan 1978), and mid- to late-May in northern Michigan (Ewert and Hamas, unpubl. data) and northern Minnesota (Ahlgren 1957). At the latitude of northern Michigan, spruce budworm (*Choristoneura fumiferana*) larvae, and perhaps other caterpillars, become available as prey at the third instar in late May to early June (Baker 1972). Thus, aquatic habitats that produce large numbers of emerging insects may be particularly important early in spring migration, especially in northern areas where there may be less overlap between migration and emergence of leaves. These habitats may also be important to breeding birds as sources of emergent insects in some areas (Busby and Sealy 1979, Biermann and Sealy 1982, Guinan and Sealy 1987).

Evidence from recent studies suggests that emerging aquatic insects are important to spring migrants. Preliminary results indicate that forested sites within 0.8 km of the Lake Ontario coastline in New York (Agard and Spellman 1994) and along coastlines of Lake Huron and Lake Superior in Michigan (Ewert and Hamas, unpubl. data; Johansen 1993) have the highest concentrations and species richness of migrants during spring migration. Attack rates by black-throated green warblers (*Dendroica virens*) and American redstarts (*Setophaga ruticilla*) on prey, mostly midges in spring, were higher along the shoreline compared to inland sites in spring and in fall (Seefelt, unpubl. data). Additional work is needed to determine if differences in attack rate correlate with changes in weight during migration. Nonetheless, these data suggest that migrants may select or remain near the shoreline as a result of a relatively abundant and perhaps predictable supply of midges emerging from the water.

When cold spring weather delayed phenological development and reduced insect activity, Sealy (1988, 1989) found that Cape May warblers (*Dendroica tigrina*) moved from trees, where they foraged on insects, to the edge of water where they fed on flying insects or defended willow catkins from conspecifics and other species. Territorial behavior has also been noted in migrant northern waterthrushes in coastal Texas wetlands (Rappole and Warner 1976) and in response to locally rich food supplies by Cape May warblers (Kale 1967) and palm warblers (*Dendroica palmarum*) (Wunderle 1978) in Florida. In extremely cold weather, even habitats near water will not sustain migrants and starvation occurs (Ligon 1968, Whitmore *et al.* 1977).

During fall migration, most insectivorous migrants in the Midwest move south in late August and early September (Bennett 1952, Brock 1992) when insect densities may be near their maximum (see Kendeigh 1979) while frugivorous and granivorous birds migrate somewhat later (Bennett 1952). Food for birds feeding off foliage may be more widely distributed in fall compared to spring, particularly in the northern Midwest. Further, at least some of these species feed on fruit in fall, a resource typically not available during spring migration (Parrish, in press). Thus, in fall, migrants may occupy a wide range of habitats (see Weisbrod *et al.* 1993), or at least select different habitats compared to spring (Winker *et al.* 1992b). This assessment, based on field observations by ourselves and others, has yet to be adequately tested in the Great Lakes region and is only partially supported by studies in other landscapes. In Arizona, for example, different species groups of migrants occupied either wider or narrower ranges of habitats in fall compared to spring, but the habitats they selected generally had relatively high insect abundance (Hutto 1985). Much additional work is needed to evaluate the relationship between food availability and the breadth and regional variation of habitat selection by migrants in the fall.

Predation pressure during migration may also affect microsite selection. Along the shores of the Great Lakes, predation on landbird migrants may be relatively high because shorelines are also migratory corridors for raptors (Peet 1908, Tyrrell 1934, Berger 1954, Weir *et al.* 1980). Predation pressure should vary seasonally along Great Lakes shorelines because raptors tend to be most abundant along eastern and southern shores of the lakes in spring and along western and northern shores in fall. Herring gulls (*Larus argentatus*)

and ring-billed gulls (*Larus delawarensis*) also prey on migrants along the shoreline (Ziarno 1994, French 1994, Kielb 1994). By selecting areas with complex vegetation structure, migrants may decrease the probability of predation by raptors and other birds (see Davidson 1994). Birds may also aggregate in flocks to avoid predation during migration (e.g., Lindstrom 1989), although it is unclear how habitat selection or social organization during migration is modified by the presence of predators.

RECOMMENDATIONS

Managing and protecting stopover sites for migratory birds should be components of conservation programs to protect and manage landscapes. For that reason, we believe protection of natural processes and native species, including the full range of non-avian species and natural communities, should ultimately guide management programs at stopover sites for neotropical migrants wherever possible. Where lands are managed, we suggest a range of successional habitats be maintained with a distribution that does not fragment habitat for area-sensitive species. We recognize that management, such as creation of early successional or perhaps aquatic habitat, may occasionally be needed for species with exacting requirements, and this could be decided on a case-by-case basis. It is not clear if this is needed for any landbird migrants in the Midwest although certainly some species, such as northern waterthrush, have relatively narrow habitat requirements during migration.

The following recommendations directed at protecting and managing sites for migrants are very tentative and broad, a reflection of our poor knowledge of stopover sites. Still, we hope the recommendations prompt interest in protection of stopover sites by landowners and encourage researchers to evaluate factors which govern habitat use by migrants. We propose the following measures for better protection of stopover sites:

1. A network of habitat patches (ideally to include riparian or other lowland habitats and upland habitats) be established throughout the Midwest. This may be especially critical in highly altered landscapes, such as urban areas, and in large expanses of agricultural land, as in portions of Iowa, Illinois, Indiana and Ohio, where relatively few patches of forest, grassland or wetlands remain. The network should include existing protected federal, state, county, non-profit, and corporat lands as well as lands owned by individuals. More research, similar to the design of Martin (1980), is needed to determine if these lands should be connected and, if not connected, how far apart the patches of habitat should be located. Habitat restoration may be needed to complete a network of stopover sites.

   Larger, more ecologically diverse tracts of land with natural mosaics of forests, wetlands, grasslands and openings, are likely to be consistently attractive to many migrant species and individuals given the complex interaction that occurs between species-specific requirements, food availability and shelter. This may be most important for spring migrants when food may be relatively scarce and locally distributed. However, habitats should be managed based on principles of natural process as much as possible and should not result in adverse effects on area-sensitive or habitat-interior species.

   Small patches of habitat that are unsuitable for breeding birds (Robinson 1996) may be acceptable to migrants, although this relationship has not been adequately tested. Consequently, small, isolated patches could be considered when implementing comprehensive conservation plans for stopover sites. However, for both large and small patches of habitat that can be protected or managed we recommend that stopover sites have as many of the following features as possible:

   a. Ecologically diverse plant communities with water that produces good numbers of emergent, aquatic insects, particularly in northern areas where phenological development of vegetation is delayed.

   b. Forest gaps, or at least areas with a well-developed shrub layer and high foliage density; these gaps may result from windthrow, fluctuating water levels, other disturbances, or carefully designed management which does not result in increased nest predation or cowbird parasitism.

c. Structurally complex and diverse vegetation layers.

2. Insect populations at the site and in immediately surrounding areas should be maintained. Use of biological control or spraying of insecticides should be avoided at critical stopover sites and in the immediate vicinity of these stopover sites whenever possible.

3. Introduction of alien or exotic plants and animals should be evaluated critically because this often results in disruption of ecological communities and perhaps loss of food supplies for migrants. Purple loosestrife (*Lythrum salicaria*), for example, supports few native animal species and displaces native plant species (see Thompson *et al.* 1987) that may be critical sources of food for birds at stopover sites.

4. Identification and protection of spring stopover sites may be especially important because migrants may be more susceptible to food limitations and climatic variability compared to fall. However, fall migrants may be subject to periodic food shortages because of drought, or other conditions, and this may be disproportionately stressful on young birds, especially at sites where competition for food and shelter is severe. Research directed at evaluating these relationships and identifying these stopover sites should be a high priority because the nature of resource limitations varies geographically and temporally.

5. Coastal areas along the Great Lakes and other large bodies of water should receive special attention because of their importance as landfall areas. Protection of water quality and subsurface bottom features should also be sought where nearshore larval insects occur. More research is needed to define which areas of shoreline, both terrestrial and aquatic, are most important to migrants during both the spring and fall migrations and should focus on the interaction between food supplies and distribution of migrants especially.

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Reviews status of Neotropical migratory landbirds and effects of land-use practices in the Midwestern United States, from a landscape perspective, through a series of papers authored by regional experts. Includes recommendations for the conservation of Midwestern Neotropical migratory landbirds.

KEY WORDS: Neotropical migratory landbirds, Midwestern United States, landscape ecology, land-management practices, grassland habitat, agricultural habitat, forest habitat, bottomland forest.

Our job at the North Central Forest Experiment Station is discovering and creating new knowledge and technology in the field of natural resources and conveying this information to the people who can use it. As a new generation of forests emerges in our region, managers are confronted with two unique challenges: (1) Dealing with the great diversity in composition, quality, and ownership of the forests, and (2) Reconciling the conflicting demands of the people who use them. Helping the forest manager meet these challenges while protecting the environment is what research at North Central is all about.